FACTORS INFLUENCING KEMP'S RIDLEY SEA TURTLE (*LEPIDOCHELYS KEMPII*) DISTRIBUTION IN NEARSHORE WATERS AND IMPLICATIONS FOR MANAGEMENT

A Dissertation

by

TASHA LYNN METZ

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

August 2004

Major Subject: Wildlife and Fisheries Sciences

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Approved as to style and content by:

André M. Landry, Jr. (Chair of Committee)	Lee A. Fitzgerald (Member)
Jane M. Packard (Member)	James W. Webb (Member)
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ABSTRACT

Factors Influencing Kemp's Ridley Sea Turtle (*Lepidochelys kempii*) Distribution in Nearshore Waters and Implications for Management. (August 2004)

Tasha Lynn Metz, B.S., Texas Christian University;
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Post-pelagic juvenile and subadult Kemp's ridley sea turtles (*Lepidochelys kempii*) (20-40 cm straight carapace length) utilize nearshore waters of the northwestern Gulf of Mexico as nursery or developmental feeding grounds. This study utilizes 10 years of entanglement netting data to characterize long-term abundance and distribution of Kemp's ridley sea turtles at index habitats in this region. Netting surveys were conducted during April-October 1993-2002, primarily at Sabine Pass, Texas and Calcasieu Pass, Louisiana. Additionally, this study takes an ecosystem-based approach to understanding factors influencing Kemp's ridley in-water abundance and distribution via the development of a conceptual model incorporating data on nesting dynamics, environmental conditions, prey availability, and predation pressure.

Overall monthly mean ridley catch-per-unit-effort (CPUE) peaked in the beginning of summer (April-June), probably in response to rising water temperatures and seasonal occurrence of blue crab prey. Annual mean ridley CPUE across all study areas peaked in 1994, 1997, 1999 and 2002, suggesting a 2-3 year cycle in abundance that may be related to patterns in clutch size or hatch success at the Rancho Nuevo, Mexico nesting beach. However, ridley CPUE in nearshore waters remained relatively constant or decreased slightly even as number of hatchlings released from Rancho Nuevo increased exponentially. Annual declines in Texas strandings since 1994 and subsequent increases in Florida counterparts since 1995 suggest a shift in ridley distribution from the western to eastern Gulf in recent years.

Significant declines in ridley CPUE at Sabine Pass since 1997 coincided with a concurrent reduction in blue crab size, but a similar trend was not detected at Calcasieu Pass. Kemp’s ridley occurrence at study sites was not significantly related to shrimping activity/by-catch. There also were no biologically significant relationships between Kemp’s ridley CPUE and abiotic factors, nor were ridleys deterred from utilizing areas frequented by bull sharks. Overall, nesting dynamics and prey availability were conceptual model components appearing to have the greatest influence on nearshore ridley occurrence.

DEDICATION

to my family and friends

who have supported me all these years…

ACKNOWLEDGEMENTS

I would like to extend my sincerest gratitude to those people who have supported me in my education and research endeavors. Without their contributions and encouragement, this achievement would not have been possible.

I would like to thank my committee members Drs. André M. Landry, Jr., Lee A. Fitzgerald, Jane M. Packard, and James W. Webb for their valuable insight and advice. I would especially like to thank Dr. André M. Landry, Jr. for serving as my committee chair and major professor. His commitment to excellence, along with his patience and understanding, inspired me to accomplish my goals to my utmost ability.

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CHAPTER I

INTRODUCTION

The Kemp's ridley, *Lepidochelys kempii* (Garman 1880), is the most critically endangered sea turtle in the world (Magnuson *et al.*, 1990; IUCN, 2003). Although this species spends over 99% of its life at sea, very few studies have assessed the dynamics of this "in-water" (as opposed to nesting beach) existence (Magnuson *et al.*, 1990; Turtle Expert Working Group [TEWG], 2000; Epperly, 2000). This study utilizes 10 years of entanglement netting survey data to characterize long-term abundance and distribution of Kemp's ridleys in nearshore waters of the northwestern Gulf of Mexico. Additionally, this study takes an ecosystem-based approach (Slocombe, 1993; Costanza and Ruth, 1998; Ferrero and Fritz, 2002) to understanding the factors influencing Kemp's ridley in-water abundance and distribution via the development of a conceptual model that incorporates aspects of nesting dynamics, environmental conditions, prey availability, and predation pressure. Information gathered by the present study is designed to aid in the management and continued recovery of this endangered species by increasing our knowledge of in-water life history stages and their habitat requirements.

In contrast to most other sea turtle species with circumglobal distribution, *L. kempii* is primarily confined to the Gulf of Mexico and United States east coast and has only one major nesting beach at Rancho Nuevo, Tamualipas, Mexico (at ~ 23° N, 97° 45' W). This species also is unique in that it nests during daylight hours in large assemblages or "arribazónes" that render it highly susceptible to human exploitation.

This dissertation follows the style and format of the Journal of Herpetology.

An estimated 40,000 ridley females were filmed nesting in a single day at Rancho Nuevo in 1947 (Hildebrand, 1963; Carr, 1963), but by the 1960s, harvest of ridleys for eggs and meat reduced the nesting population to about 2000 females per arribazón (Márquez, 2000). This drastic decline in "nesters" prompted the Mexican government to protect the nesting beach with armed marines beginning in 1966. Nests also were relocated to a fenced corral for greater protection from poachers and natural predators. In 1978, a bi-national team of scientists from Mexico and the US was formed to monitor the nesting population via counting nests and tagging females (Márquez, 1994). Despite these conservation measures, a record-low number of females (~350) nested in 1985.

The overlap of Kemp's ridley foraging habitat with areas of intense commercial shrimping effort (e.g. the northwestern Gulf) contributed to this species' continued population decline. Incidental capture and drowning of ridleys in shrimp trawls impacted recruitment to the nesting population and led to record low nesting activity (740-752 nests/season) during 1985-1987 (Márquez *et al.*, 1999). In 1989, the National Marine Fisheries Service (NMFS) called for voluntary use of Turtle Excluder Devices (TEDs) in commercial shrimp trawls, and instituted mandatory compliance by 1994 to prevent continued shrimping-related mortalities. These conservation measures by the US and Mexican governments have resulted in an increase in the nesting population to approximately 3000 females in 2002, a level considered indicative of a modest recovery. The Kemp's ridley downlisting criterion of 10,000 nesting females by year 2020 (United States Fish and Wildlife Service [USFWS] and NMFS, 1992) remains attainable as long as present rates of population increase continue (average of 11.3% more nests per year during 1985-1999) (TEWG, 2000). However, restricting the management focus to nesting dynamics and/or incidental capture in the shrimp fishery overlooks other potential threats to Kemp's ridley survival. Additional information on factors affecting all ridley life history stages is essential for effective management and achieving long-term recovery goals.

Kemp's ridleys and other sea turtle species are long-lived, slow-maturing animals that follow a similar general life cycle (Fig.1) consisting of hatchling, pelagic post-hatchling, coastal-benthic immature and coastal-benthic adult life history stages (Magnuson, *et al.*, 1990, Miller, 1997; Musick and Limpus, 1997). *L. kempii* is the smallest sea turtles species, with an average adult size of 60-70 cm straight carapace length (SCL), and shorter duration of each life history stage compared to other species (Márquez, 1994). Most research has focused on nesting constituents (i.e. eggs, hatchlings and nesting females) primarily due to greater accessibility of beach locations and historical use of nesting parameters as indicators of population status. In-water studies are more logistically difficult because they require extensive hours at sea to locate or capture turtles in a vast aquatic environment. However, monitoring young, in-water life history stages may give managers advance warning of changes in population abundance that impact future reproductive success and population growth (Epperly, 2000). Crouse *et al.* (1987) demonstrated via a Lefkovitch stage-based model that increased survival of juveniles and subdaults was more significant in promoting loggerhead (*Caretta caretta*) population growth than was protection of eggs and

Sea Turtle Life Cycle

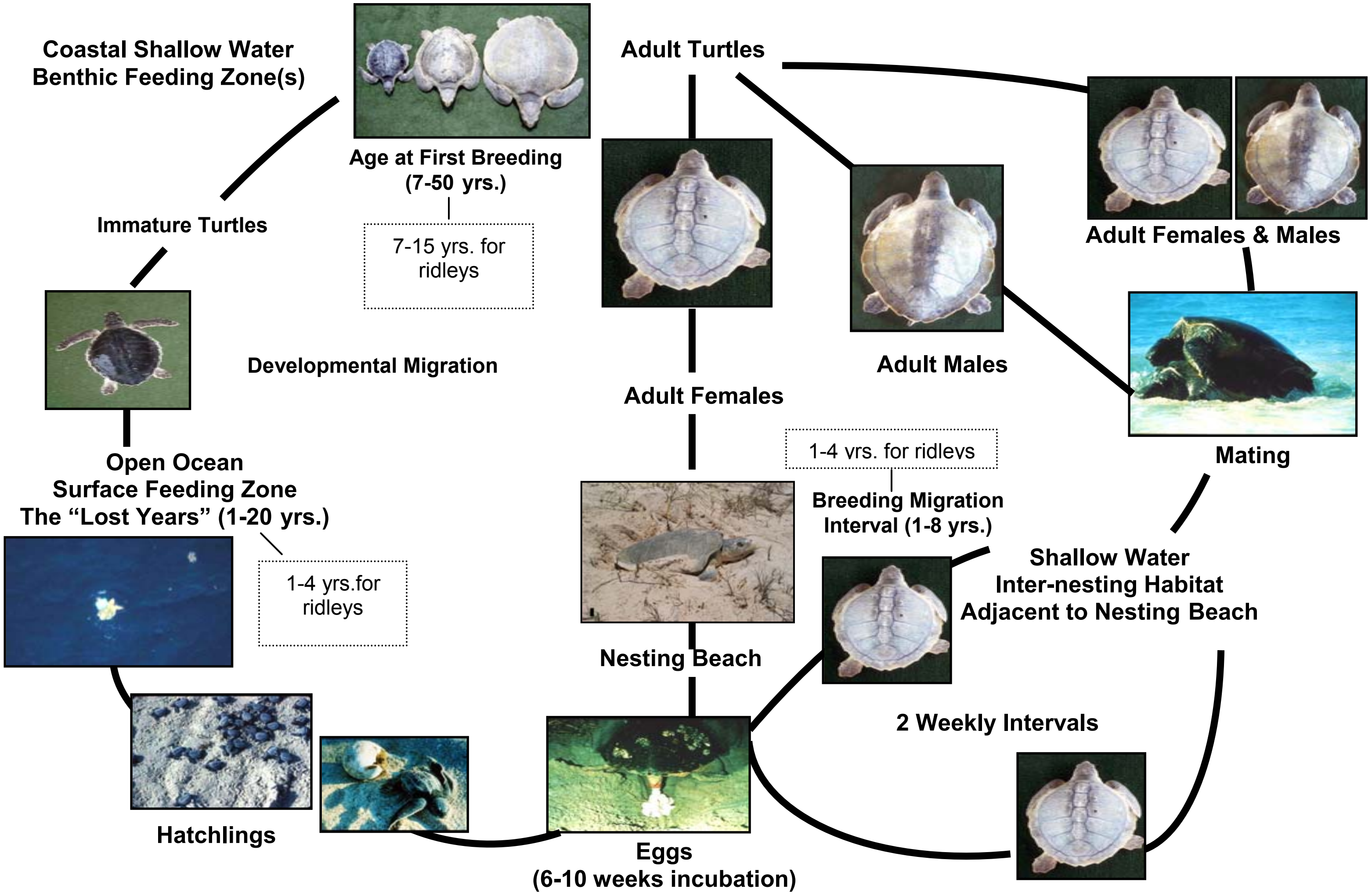


Figure 1. Generalized sea turtle life cycle showing various life history stages. Adapted from Miller (1997). Note: Dotted boxes represent life stage duration estimates for ridleys.

hatchlings due to the higher reproductive value of large immature turtles. There is a lack of information on other factors, such as habitat quality (i.e. prey availability and pollution) in nearshore nursery or “developmental” [term used to describe areas primarily used by immature turtles (Musick and Limpus, 1997)] feeding grounds that may affect their survival and fitness. The Turtle Expert Working Group (2000) has developed an age-based, deterministic model of Kemp’s ridley population dynamics, but the model is questionable, due, in part, to insufficient data on juvenile ridley survivorship. Furthermore, little is known about the habitat requirements and long-term abundance patterns of coastal-benthic immature ridleys that may be useful in understanding the ecology and survival of these in-water life stages.

The Sea Turtle and Fisheries Ecology Research Lab (STFERL) at Texas A&M University-Galveston has been conducting in-water entanglement netting surveys at Kemp’s ridley historical “index habitats” [locations that have a consistent occurrence of constituent life stages (juvenile through adult) (Landry and Costa, 1999)] along the Texas and Louisiana coasts since 1992. This 10-year dataset is the longest of its kind in the northwestern Gulf of Mexico and provides valuable information on long-term population status (i.e. abundance, distribution, and size composition) and habitat use patterns for ridleys in nearshore foraging habitat. Research conducted herein utilizes this dataset to assess factors influencing Kemp’s ridley in-water occurrence and is facilitated by the development of a conceptual model.

Conceptual Model

A conceptual model is a qualitative representation of components used to define a system of interest (Grant, 1986; Jackson *et al.*, 2000). Conceptual models are particularly useful as a first step in developing mathematical or predictive models because they provide a framework for gathering information and testing hypotheses regarding relationships between system components (Jackson *et al*., 2000; Ferrero and Fritz, 2002). The process of formulating a conceptual model involves: 1) bounding the system of interest; 2) identifying components of the system and connections between components; and 3) formally displaying the model (Grant, 1986). The conceptual model presented herein represents an ecosystem-based approach, such as was developed for the management of endangered Stellar sea lions (*Eumetopias jubatus*) in Alaskan waters (Ferrero and Fritz, 2002). The system of interest for my conceptual model focuses on Kemp's ridley occurrence in nearshore waters off the upper Texas and Louisiana coasts, an area of important foraging grounds for immature Kemp's ridleys and, occasionally, adult females (Hildebrand, 1982; Ogren, 1989; Manzella and Williams, 1992, Renaud *et al.*, 1996; Landry and Costa, 1999).

Identification of components in the conceptual model is based on ecological principles and known aspects of Kemp's ridley biology. These major model components include nesting patterns, environmental conditions, prey availability (blue crabs), and predation pressure (bull sharks). Because the nesting beach at Rancho Nuevo is the primary source of juvenile and subadult ridleys, patterns of nesting productivity may explain trends in recruitment potential to nearshore foraging habitat. In turn, rate of recruitment into the breeding population is a function of juvenile survival. Environmental conditions presumably affect ridley use of nearshore habitat on a seasonal or annual basis, as well as this species' distribution across and within regions. Abiotic factors, such as water temperature, salinity and visibility may significantly affect this species' nearshore occurrence via direct (physiological tolerance) or indirect (effects on prey or predators) mechanisms. Kemp's ridleys primarily feed on crabs, with prey species consumed differing between regions (Hildebrand, 1982; Ogren, 1989; Shaver, 1991; Burke *et al.*, 1994; Werner, 1994). Werner (1994) reported that blue crab (*Callinectes sapidus*) was the dominant species in fecal samples from wild ridleys in the NW Gulf. Because immature ridleys utilize nearshore waters as foraging grounds, availability or quality of the blue crab resource may significantly influence ridley habitat selection, and/or duration in respective habitats. However, the threat of predation may deter ridleys from foraging in a particular area even if prey availability is favorable (Lima and Dill, 1990; Krebs and Davies, 1993). One likely predator of *L. kempii* in nearshore Gulf waters is the bull shark (*Carcharhinus leucas*) due to its co-occurrence with ridleys in shallow coastal habitats and opportunistic feeding behavior (sea turtle remains have been found in bull shark stomach contents) (Clark and von Schmidt, 1965; Branstetter, 1981; Castro, 1983; Compagno, 1984; Snelson *et al.*, 1984; Grace and Henwood, 1997; Shipley, 2000).

A "box and arrow" diagram (Jackson *et al.*, 2000) is shown in Figure 2 to illustrate the connections between components of the conceptual model presented herein.

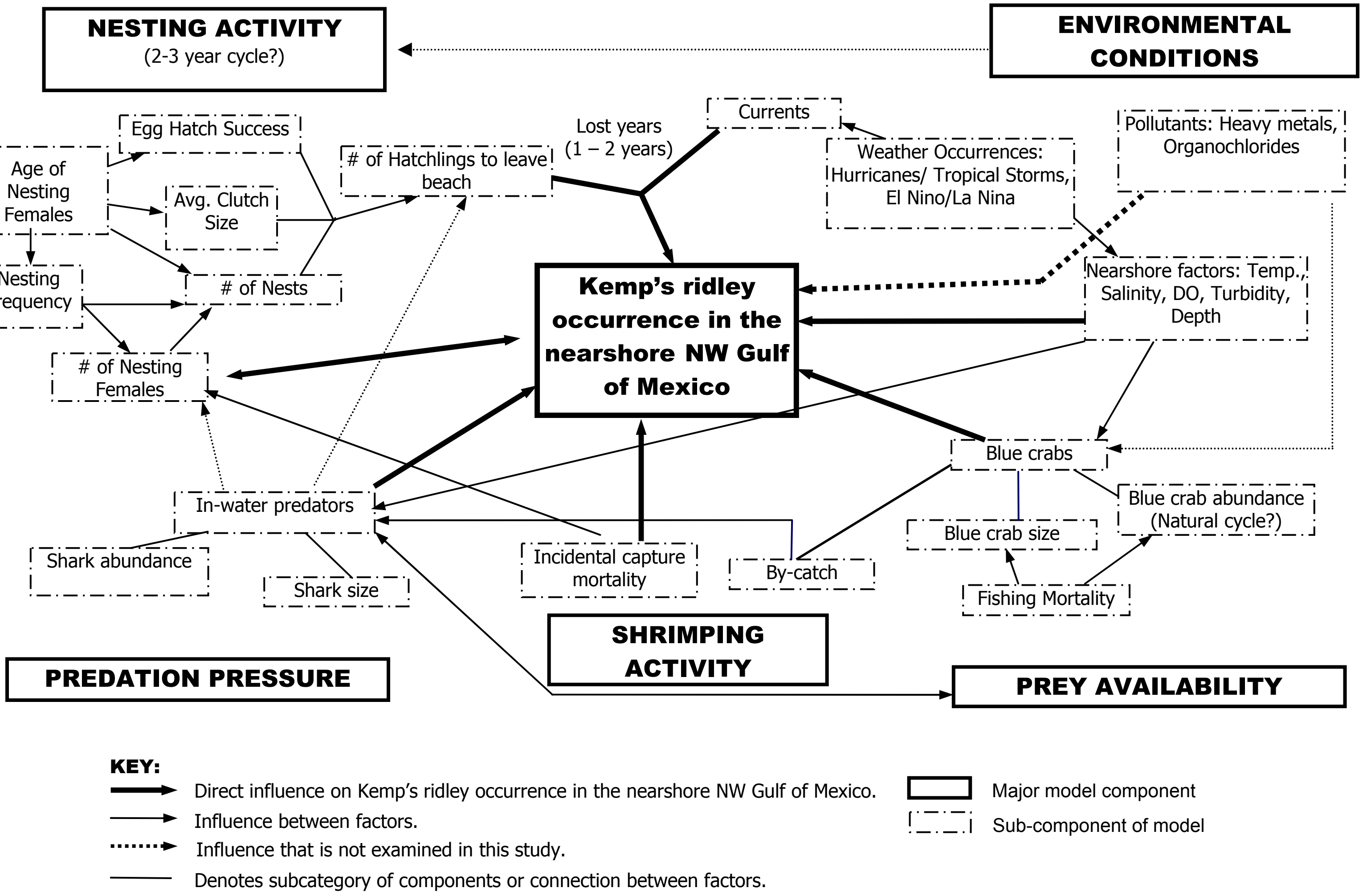


Figure 2. Working schematic of the conceptual model for factors influencing Kemp's ridley occurrence in nearshore waters of the NW Gulf of Mexico.

Table 1 describes the hypothesized influence of each component and accompanying data sources for analyses. This conceptual modeling approach is not only useful for identifying factors that may influence Kemp's ridley habitat use, but it also plays a role in evaluating the robustness of each information base (i.e. component). Ultimately, the goal of this conceptual model is to provide information and generate questions upon which future in-water research and management or a predictive model may be based.

Research Objectives

Data evaluated by this study are designed to aid in the management and continued recovery of the critically endangered Kemp's ridley by identifying which hypothesized factors have a significant relationship with long-term patterns of in-water occurrence across nearshore habitats of the northwestern Gulf. The following research objectives were established to accomplish this task:

1) To characterize Kemp's ridley size, abundance and distribution, as well as factors hypothesized to influence these parameters, at Sabine Pass, Texas, Calcasieu Pass, Louisiana, and across these sites combined in the NW Gulf.
2) To test hypotheses regarding the relationship between Kemp's ridley estimated abundance at selected index habitats and major components of the conceptual model: nesting activity, abiotic factors, prey availability and predation pressure.

Table 1. Summary of hypothesized factors and effects on Kemp's ridley occurrence in the northwestern Gulf of Mexico, including sources of data for analysis.

A. Nesting Parameters		
Factor	Possible Effect	Data Sources
Age of nesting females	Influences fecundity of nesting females, nesting frequency, egg viability/hatchling survival and clutch size. Typically neophyte nesters have smaller clutches, reduced nesting frequency and lower hatchling survival rate as compared to reimmigrants.	- Kemp's Ridley Expert Working Group Reports
Number of nesting females Number of nests Clutch size	Influences potential number of hatchlings. It is estimated that each female nests 2.3 times per season. There is evidence that 20% of ridleys nest every year, 60% every 2 years, 15% every 3 years, 5% every 4 years (TEWG, 2000)	
Potential number of hatchlings to leave the beach Hatchling Survival	Influences potential number of juveniles and subadults that recruit to developmental feeding grounds.	
Natural nesting cycle	Possible 2-3 year cycle in nesting activity due to nesting fecundity and re-migration interval.	
B. Environmental Conditions		
Weather patterns	Can change or disrupt currents and thus the transport of hatchlings, as well as influence abiotic factors in nearshore waters, which in turn, impact prey dynamics.	- National Weather Service
Gulf circulation and currents	Transport hatchlings from nesting beach to pelagic environment and, later, to feeding grounds.	- Literature review - Sea Surface Height maps 1993-2002 from the Colorado Center for Astrodynamis Research
Nearshore Abiotic conditions (e.g. temperature, salinity, dissolved oxygen, turbidity and depth)	Affects habitat quality for Kemp’s ridley. Affects Kemp's ridley distribution by influencing distribution and abundance of predators and prey.	- Literature review - Sea Turtle and Fisheries Ecology Research Lab (STFERL)
C. Prey availability		
Blue crab abundance Blue crab size Natural life history cycle	Influences ridley distribution by affecting foraging success. Ridleys may encounter prey more often when foraging in areas of abundant crab stocks, and may influence prey selection, if it exists. Greater foraging success could lead to increased growth and earlier sexual maturation on the part of ridleys, as well as, increased duration on feeding grounds. However, this may also result in increased susceptibility to encounters with the shrimp fishery.	- STFERL - Texas Parks and Wildlife Dept. (TPWD)

Table 1 (cont.). Summary of hypothesized factors and effects on Kemp's ridley occurrence in the northwestern Gulf of Mexico, including sources of data for analysis.

C. Prey availability (cont.)		
Blue crab fishery	Overexploitation may indirectly affect ridley distribution by decreasing the size (juvenescence) and abundance of crab stocks available as prey.	- TPWD monitoring reports and commercial fishing licenses - Literature review
Shrimp Fishery By-catch	Influences ridley distribution because discarded crabs and other items are consumed by foraging Kemp's ridleys.	- STFERL (Sparks, 1999) - TPWD monitoring reports and commercial fishing licenses - National Marine Fisheries Service (NMFS) - Sea Turtle Stranding and Salvage Network (STSSN)
D. Shrimping Activity		
Shrimping by-catch	Increases food availability and potential acquisition by clumping the distribution of crabs and other items discarded en-mass from shrimp boats.	- Literature review
Incidental capture mortality	Reduces the number of ridleys and usually coincides with areas of high shrimping activity.	- STSSN - NMFS
E. Predators		
In-water predators: Shark abundance Shark size	May deter ridleys from foraging in a particular area due to risk of predation. May also compete with ridleys for food.	- Literature review - STFERL data (Brooke Shipley's thesis, photo evidence of bite marks) - TPWD monitoring data
Prey availability and by-catch	Indirectly affects ridleys and influences shark distribution because juvenile bull shark feeding grounds overlap with ridleys.	

Hypotheses

The following hypotheses, as well as the above research objectives, conceptual model components, and methods for testing hypotheses, are more fully addressed in subsequent chapters:

1) Kemp's ridley abundance at study areas will be significantly correlated with number of hatchlings released from the Rancho Nuevo nesting beach and patterns in nesting activity.
2) Kemp's ridley occurrence at study areas will be positively correlated with water temperature and salinity.
3) Kemp's ridley occurrence at study areas will be associated with the abundance and size of blue crab prey.
4) Kemp's ridley occurrence at study areas will be negatively correlated with bull shark abundance and distribution.

CHAPTER II

KEMP’S RIDLEY DYNAMICS

Introduction

Effective management of Kemp’s ridley sea turtle population recovery necessitates a greater understanding of factors influencing this species’ in-water occurrence and survivorship (USFWS and NMFS, 1992). To date, most ridley conservation measures have focused on nesting beach protection and reduction of incidental capture in commercial shrimp trawls. While these efforts have seemingly contributed to an increase in nesting activity at Rancho Nuevo, Mexico, there is still a lack of information on abundance, distribution, and habitat requirements of in-water life history stages. Furthermore, the connection between ridley abundance in nearshore habitat and patterns of nesting activity at Rancho Nuevo has not been well documented. This void in our knowledge of Kemp’s ridley dynamics makes it difficult to adequately understand this species’ ecology. As such, a conceptual model has been developed as a first step in identifying and evaluating the influence of various hypothesized factors on Kemp’ ridley occurrence in developmental feeding grounds of the northwestern Gulf of Mexico (Fig. 2). This chapter contributes to this assessment by first characterizing Kemp’s ridley size, abundance and distribution at Sabine Pass, Texas and Calcasieu Pass, Louisiana, as well as across all sites combined in the NW Gulf via 10 years of entanglement netting data. In addition, this chapter assesses the relationship of ridley occurrence in the NW Gulf with nesting productivity at Rancho Nuevo (1978-2002) (Fig. 3). Lastly, ridley stranding statistics (with consideration of trends in commercial

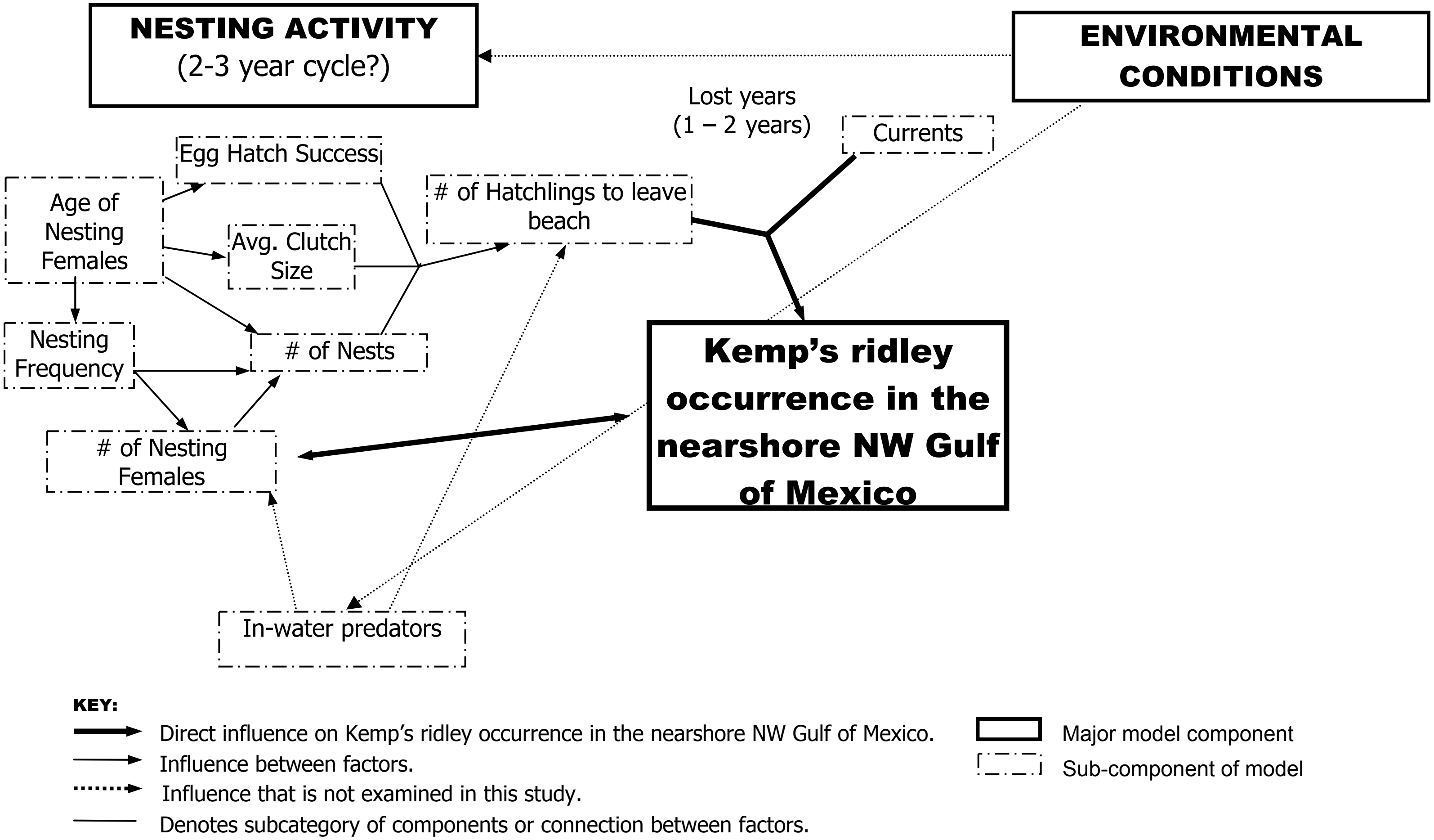


Figure 3. Portion of conceptual model detailing the hypothesized influence of nesting factors on Kemp's ridley occurrence in nearshore waters of the NW Gulf of Mexico.

fishing effort) from the western Gulf (Texas; 1994-2002), eastern Gulf (Florida; 1987-2002) and east coast (North Carolina; 1993-2002) are examined to provide additional geographic information on ridley occurrence within US coastal waters.

In-water Occurrence

In-water captures, stranding surveys and tracking studies indicate that immature Kemp's ridley sea turtles (20-55 cm SCL) primarily inhabit nearshore waters of the Gulf of Mexico and US east coast (extending as far north as Massachusetts during summer) (Liner, 1954; Dobie *et al.*, 1961; Carr, 1977; Lazell, 1980; Hildebrand, 1982; Lutcavage and Musick, 1985; Henwood and Ogren, 1987; Byles, 1989; Ogren.1989; Márquez, 1990; Rudloe *et al.*, 1991; Manzella and Williams, 1992; Burke *et al.*, 1994; Márquez, 1994; Schmid, 1995; Landry and Costa, 1999). In rare instances, Kemp's ridleys carried by the Gulf Stream enter the North Atlantic gyre and have been found in England, France, the Mediterranean and Nova Scotia (Brongersma, 1972; Manzella *et al.*, 1988). It was once speculated that ridleys found along the US Atlantic coast were waifs carried on currents through the Florida straits and lost to the breeding population (Carr, 1980; Hendrickson, 1980; Magnuson *et al.*, 1990). However, through tagging efforts, we now know that some of these immature ridleys return to the nesting beach (Witzell, 1998).

Most young ridleys are found in the northern Gulf from Texas to Florida, particularly along the upper Texas/Louisiana coast and near Cedar Key, FL, because of this region's proximity to the nesting beach and abundant prey (Hildebrand, 1982; Ogren, 1989). Manzella and Williams (1992) examined 865 records of *L. kempii* occurrence along the Texas coast and found the highest frequencies concentrated in the "Sabine/High Island", "Galveston/Bolivar Roads" and "Corpus Christi Bay/North Padre Island" regions.

In-water capture statistics indicate highest seasonal abundance of Kemp's ridleys in nearshore waters occurs during April to August, followed by sharp declines from November through March due to conspecifics moving into deeper, warmer waters with onset of cooler water temperatures (Renaud *et al.*, 1995; Landry and Costa, 1999). Seasonal occurrence and movements of ridleys near Cedar Key, FL and along the east coast mirrors that of western Gulf conspecifics (Lazell, 1980; Lutcavage and Musick, 1985; Henwood and Ogren, 1987; Byles, 1989; Ogren, 1989; Burke *et al.*, 1994, Schmid, 1995). Entanglement net surveys and telemetric tracking of ridleys released near Sabine Pass, TX and Calcasieu Pass, LA demonstrated that smaller individuals (<18 kg) exhibit strong site fidelity to tidal passes presumably because they are attracted to high blue crab abundances that occur within 4.9 km of land and 20 km of their release site. In contrast, larger ridleys (>24 kg) are more migratory, traveling > 17 km from shore and as far as 2600 km after release (Renaud *et al.*, 1995; Landry and Costa, 1999).

Although previous research has been valuable in determining Kemp's ridley's overall range of occurrence and short-term habitat use, few studies have examined long-term abundance trends or distribution patterns. Long-term trends in coastal-benthic Kemp's ridley abundance are best assessed via prolonged monitoring surveys. Common in-water sampling techniques for sea turtles include entanglement netting, trawling, pound netting, strike netting, rodeo, and aerial surveys (Bjorndal and Bolten, 2000). The three latter techniques are not as useful at locating or capturing *L. kempii* in the north/northwestern Gulf of Mexico due to this species' relatively small size and the region's low visibility conditions. Trawling surveys have been used successfully in areas of high loggerhead densities, such as Cape Canaveral, FL (Butler *et al.*, 1987), or in concert with by-catch studies conducted onboard commercial shrimping vessels (Epperly *et al.*, 2002). However, trawling has not been commonly used as a capture means targeting ridleys in the western Gulf due to this species' patchy distribution and reduced population abundance. Entanglement netting is most useful in areas of calm seas and high turbidity, and thus has been employed regularly in nearshore waters of the northwestern Gulf for monitoring ridley abundance (Landry and Costa, 1999). Capture data can be expressed as catch-per-unit-effort or CPUE (in this case, # turtles/km-hour of netting), and, as such, takes sampling effort into account to provide a standardized measure of ridley abundance trends across sites and years.

Temporal Nesting Patterns

The abundance of immature ridleys potentially able to recruit to coastal benthic habitats is intrinsically linked to reproductive output, nesting success, and hatchling survival. Number of nests, number of eggs and number of hatchlings released from Rancho Nuevo have been quantified. Yet, several other aspects of ridley nesting biology influence these values including: number and age of nesting females; number of nests per female per season (i.e., nesting frequency); re-migration interval (i.e., time span between nesting seasons) (Miller, 1997); number of eggs per nest (i.e., clutch size); nest environmental conditions (e.g., moisture, temperature); and hatch success (i.e., egg to hatchling survival). Number of nests may not only be affected by the number of nesters but also number of nests per female per season. Márquez *et al.* (1982) estimated that adult ridley females lay an average of 1.3 nests per season, but Pritchard (1990) reported 2.3 nests/female/season after re-examining data from the 1989 nesting season. Rostal *et al.* (1997) further amended estimated nesting frequency to approximately 3 nests/female/season based on serum testosterone levels and ultrasonography. Due to the lack of consensus on the most accurate value for this parameter, the TEWG (2000) adopted 2.5 nests/female/season in its deterministic model of population dynamics because it is closest to the mean of estimates from previous studies (2.4 nests/female/season). Adult female remigration interval also may contribute to inter-annual variability in number of nests. Tagging results have indicated roughly 20% of females nest every year, 60% every 2 years, 15% every 3 years, and 5% every 4 years (TEWG, 2000).

Total number of eggs laid at Rancho Nuevo per season is dependent upon number of nesters, nests and clutch size of each nest. Adult female ridleys average about 100 eggs per nest, but older females typically produce larger clutches with higher hatching success than do neophyte nesters (Márquez *et al.*, 1989). The number of hatchlings successfully emerging from the nest also is influenced by conditions in the nest environment. Thus, annual abundance patterns of coastal-benthic immature ridleys may be significantly related to number of hatchlings (assuming that annual survivorship of ridleys as they make their way from the pelagic to the neritic zone remains relatively constant).

Stranding Statistics

Strandings data compiled by the Sea Turtle Stranding and Salvage Network (STSSN) may be utilized as supplemental observations of Kemp's ridley occurrence within and across regions (Rabalais and Rabalais, 1980; Lutcavage and Musick, 1985; Manzella and Williams, 1992; TEWG, 2000). Although stranding statistics are frequently used as a measure of sea turtle mortality (Epperly-unpublished report, 2000), they also can provide some indication of the species' abundance and distribution so long as factors affecting stranding rates and drifting of drowned sea turtles are taken into consideration (Henwood and Shah - unpublished report, 1995). Incidental capture of ridleys in commercial fishing gear has been identified as a major contributor to elevated stranding numbers (Henwood and Stunz, 1987; Magnuson *et al.*, 1990; Caillouet *et al.*, 1991, Caillouet *et al.*, 1996), especially in the western Gulf (NMFS statistical sub-areas13-21) where commercial shrimping effort is consistently high (Nance, 1993; TEWG, 2000). Wind speed/direction, surface currents, distance from shore and decomposition rate also affect where and if sea turtles strand. Even though commercial fishing effort, gear type and oceanographic conditions are not uniform across coastal regions, strandings within a particular area may still be representative of abundance patterns if fishing effort and other factors in that region remain relatively constant over time. As such, examining Kemp's ridley stranding trends in relation to commercial fishing effort may provide a complementary characterization of this species' nearshore distribution along the US coast.

This chapter characterizes Kemp's ridley size, abundance and distribution at nearshore study sites in the NW Gulf and assesses the relationship of ridley occurrence at these sites with nesting productivity at Rancho Nuevo, Mexico. It is hypothesized that abundance of immature ridleys in nursery habitats will be significantly correlated with patterns in nesting activity. In addition, ridley stranding statistics from Texas, Florida and North Carolina are examined for an additional measure of trends in Kemp's ridley abundance and distribution along the US coast.

Materials and Methods

Study Areas

In-water capture operations were conducted primarily at Sabine Pass, Texas and Calcasieu Pass, Louisiana, but secondarily near the Mermentau River, LA (the latter only sampled during 1999 and 2000), with all three sites considered nearshore nursery foraging grounds of the Kemp's ridley (Fig. 4). Sabine Pass forms the southernmost border between Texas and Louisiana, with Calcasieu Pass located 46.3 km to the east in Cameron Parish, LA. The Mermentau River also is located in Cameron Parish, about 22.4 km east of Calcasieu Pass. This proximity and similarity in shore type (i.e., saltmarsh as opposed to sandy mudflat shores at Sabine Pass) justified pooling ridley capture statistics and related observations from the Mermentau River with those from

Calcasieu Pass to supplement data analyses (referred to collectively as just “Calcasieu Pass”).

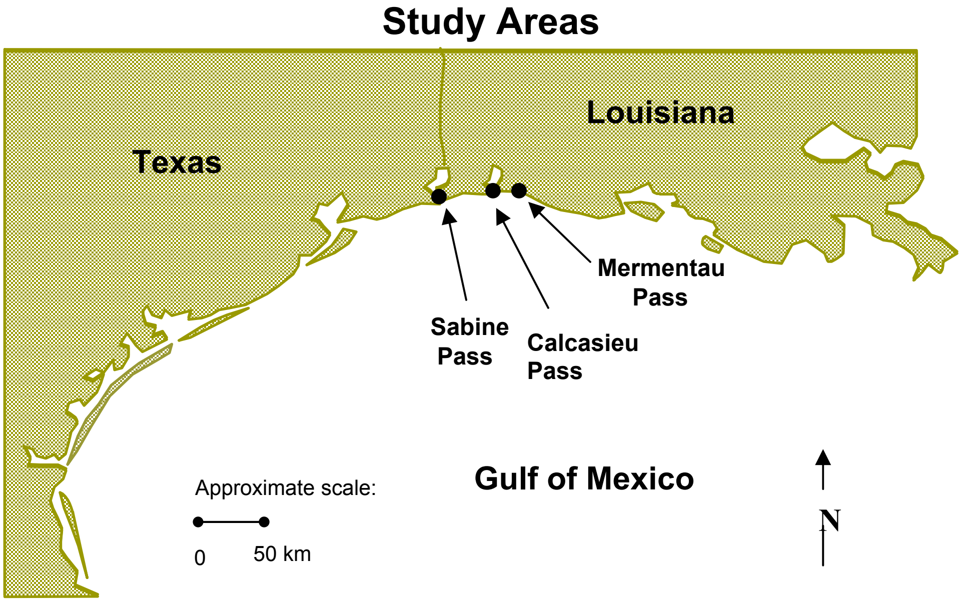


Figure 4. In-water survey locations for Kemp’s ridley sea turtles in the NW Gulf of Mexico.

Sabine Pass is bordered on the east and west by 5.6 km long granite jetties, near which four sea turtle monitoring stations have been established since 1992 (Fig. 5). Jetty stations 1 and 4 were adjacent to and gulfward of the west and east jetties, respectively, at approximately 1200-1500 m from shore. Beachfront stations 3 and 5 were on the west, respectively, and within 1 km of the jetties and 300-800 m from shore. Water depth at the jetty sites varied between 1.5 and 3.0 m, while that at beachfront sites ranged from 0.6 to 2.0 m. Bottom type consisted of a soft muddy/clay substrate at stations 1 and 3 while a more compacted sandy/mud bottom characterized eastern counterparts.

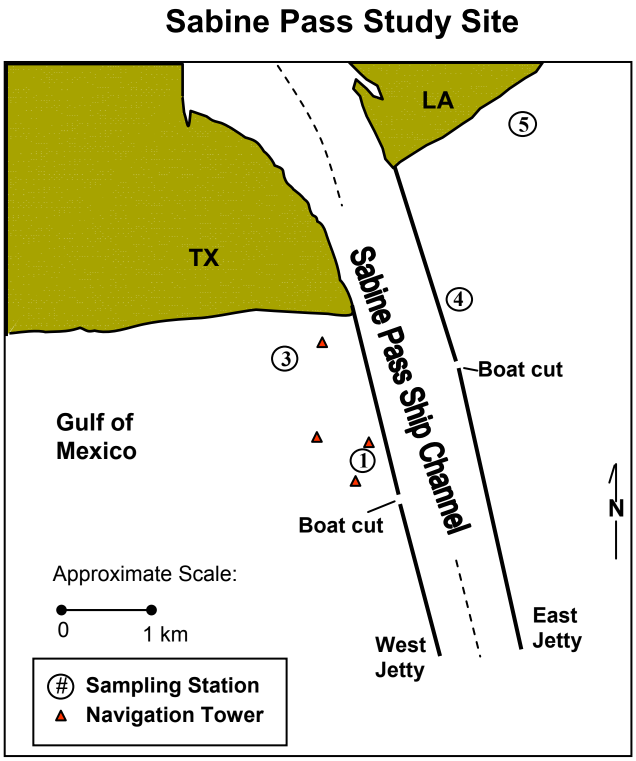


Figure 5. Map of Sabine Pass study site showing general location of sampling stations.

The nearshore topography and position of jetties at Calcasieu Pass were similar to those at Sabine Pass, with the exception of shorter jetties (approximately 3 km) protecting Calcasieu Pass (Fig. 6). Sampling stations at Calcasieu Pass mirrored those at Sabine Pass in terms of number, designation, and relative location. Only stations west of the jetties were sampled at the Mermentau River location, due to nearshore topography creating deeper water and unfavorable sampling conditions on the east side (Fig. 7).

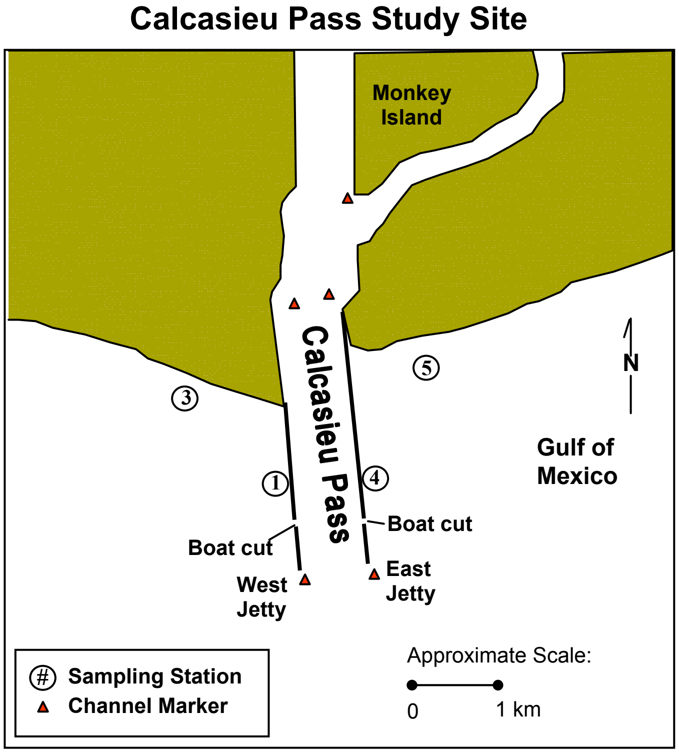


Figure 6. Map of Calcasieu Pass study site showing general location of sampling stations.

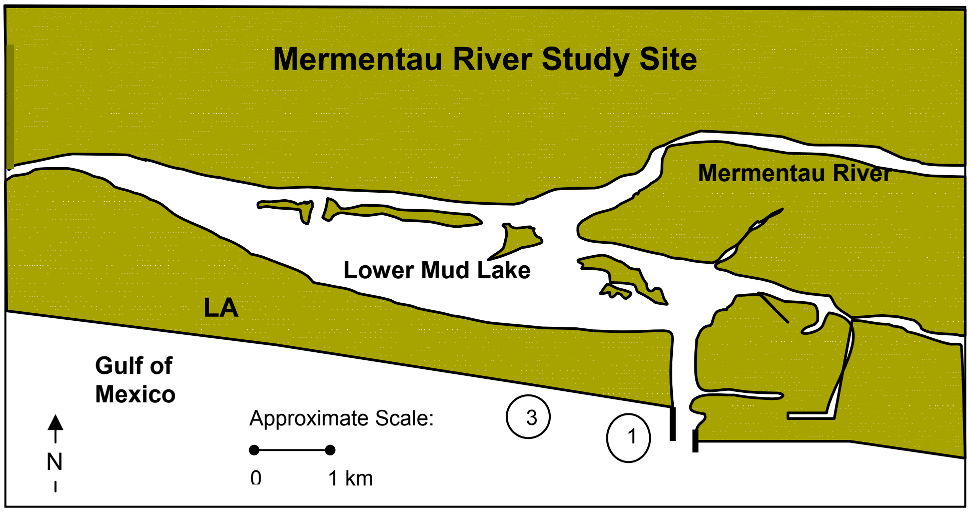


Figure 7. Map of Mermentau River study site showing general location of sampling stations.

In-water Surveys

Seasonal occurrence and abundance (expressed as catch-per-unit-effort or CPUE) of Kemp's ridleys were assessed during April – October 1993-2002 via entanglement netting operations conducted. All months were not sampled every year, and Calcasieu Pass was not sampled in 1996 and 1997. Entanglement nets were 91.4 m in length, but of different specifications: 1) 3.7 m deep with 12.7-cm bar mesh of #9 twisted nylon; or 2) 4.9 m deep with 25.4-cm bar mesh of #9 twisted nylon. Water depth and current dictated net type used at a particular station. All stations were sampled with 2-6 nets set adjacent to one another and perpendicular to the beachfront or jetty for 6-12 hours per day. Typically, one boat with 1-4 observers was responsible for monitoring 2 nets (~ 182 m of net) that were checked for sea turtles and by-catch every 20 minutes (from the end of the previous check). In addition, observers constantly watched for splashes or other signs of turtle capture to prevent or minimize risk of ridleys drowning while entangled. Pinger devices emitting high-frequency sounds at regular intervals were attached to nets to alert bottlenose dolphins (*Tursiops truncatus*) to the obstacle and reduce the possibility of incidental capture.

Sampling stations were often not selected in a random fashion because sea state, weather conditions, and water depth dictated where nets could be successfully deployed on any given day. Also, in some years the primary objective during netting surveys was to capture the most ridleys possible for individual research projects. This resulted in Gulf waters west of the jetties being preferred for netting due to these stations seemingly yielding a higher capture rate. Stations eastward of jetties were sampled primarily when conditions at western counterparts were unfavorable, and, as such, turtle occurrence at the former may be under-represented. A minimum of 3 sample days per month was targeted for each study area (Sabine, Calcasieu and Mermentau), but scheduling conflicts, weather conditions and equipment problems sometimes negated netting in a particular area or given month. This created gaps in the netting database of various months, sites or years.

Captured turtles were taken to an onshore holding facility and allowed to acclimate overnight. Morphometric characteristics of each ridley, including straight and curved carapace length (cm) and weight (kg), were recorded within 24 hours post-capture. Turtles also were inspected for evidence of being recaptured or headstart individuals via the presence of flipper tags/scars, living tags, PIT tags and wire tags. Written and photographic records were used to document the condition of captured ridleys and visible injuries. An inconel style 681 tag issued by the NMFS SEFSC-Miami was affixed to the trailing edge of each front flipper while a PIT tag was embedded in surficial tissue of the right front flipper prior to each ridley's release. Captured turtles were held for a maximum of 72 hours and then released at their capture location. An annual report of tagged turtles and recaptures was submitted to the Archie Carr Center for Sea Turtle Research (ACCSTR), which manages tagging data and facilitates the exchange of tag information in conjunction with the NMFS SEFSC-Miami. This tagging procedure was conducted in compliance with the Cooperative Marine Turtle Tagging Program (CMTTP), so that STFERL researchers as well as other agencies could identify recaptured turtles.

Nesting Data

Nesting data were obtained from annual reports prepared by the bi-national team of Mexican and US researchers monitoring the beach at Rancho Nuevo, Mexico and presented to the Kemp's Ridley Working Group (KRWG). These data include number of nests, number of hatchlings and number of eggs recorded annually since 1978. Average annual clutch size was estimated by dividing the number of eggs by number of nests, and number of nesting females was calculated by dividing the number of nests by 2.5 nests per female per season (based on estimates of nesting frequency reported by Pritchard, 1991 and Rostal, 1997; and used by the TEWG, 2000).

Strandings Data

Strandings from Texas, Florida and North Carolina, representing the western Gulf, eastern Gulf, and east coast, respectively, were used to provide additional information on Kemp's ridley distribution along the US coast. Strandings from the northeastern US were not included in analyses due to the prevalence of cold stunning events that can produce an artificially high number of strandings. Texas stranding statistics during 1994-2001 were obtained from data provided to the KRWG by state STSSN coordinator Dr. Donna Shaver-Miller (United States Geological Service (USGS) – Padre Island National Seashore). Dr. Allen Foley of the Florida Marine Research Institute provided a summary of Florida strandings via personal communication, while North Carolina strandings data were obtained from Matthew Godfrey and Wendy Cluse of the North Carolina Wildlife Resources Commission. Factors affecting stranding rates also were considered when using these data as a measure of ridley distribution.

Shrimping Effort

Kemp's ridley strandings in the Gulf of Mexico have been significantly correlated with shrimping intensity, while east coast (NC) strandings have been related to other fisheries that utilize gillnets to catch southern flounder (*Paralichthys lethostigma*) and monkfish (*Lophius americanus*). Therefore, commercial shrimping effort data for the Gulf of Mexico (since 1990) were obtained from Dr. James Nance at the NMFS-SEFSC in Galveston to assess this fishery's impact on stranding rates, primarily in Texas and Florida. Statistical sub-areas along the Gulf coast are shown in Figure 8. During 1990-2001, Sub-Areas 18-21 (entire Texas coast) had significantly higher shrimping effort ($p < 0.001$) than did counterparts in the eastern Gulf [$p = 0.01$ vs. Sub-Areas 1-9 (Florida Gulf coast); $p = 0.009$ vs. zones 10-12 (Florida Panhandle to Mississippi River)]. Although annual shrimping effort has fluctuated since 1990, no significant trend within any of the Gulf regions was detected (all trendline slopes > 0; all $p > 0.05$) (Fig. 9). Therefore, changes in annual ridley stranding statistics for Texas and Florida are probably not due to significant changes in shrimping effort, but instead may be representative of in-water abundance patterns (assuming uniform observer effort and environmental conditions over the years).

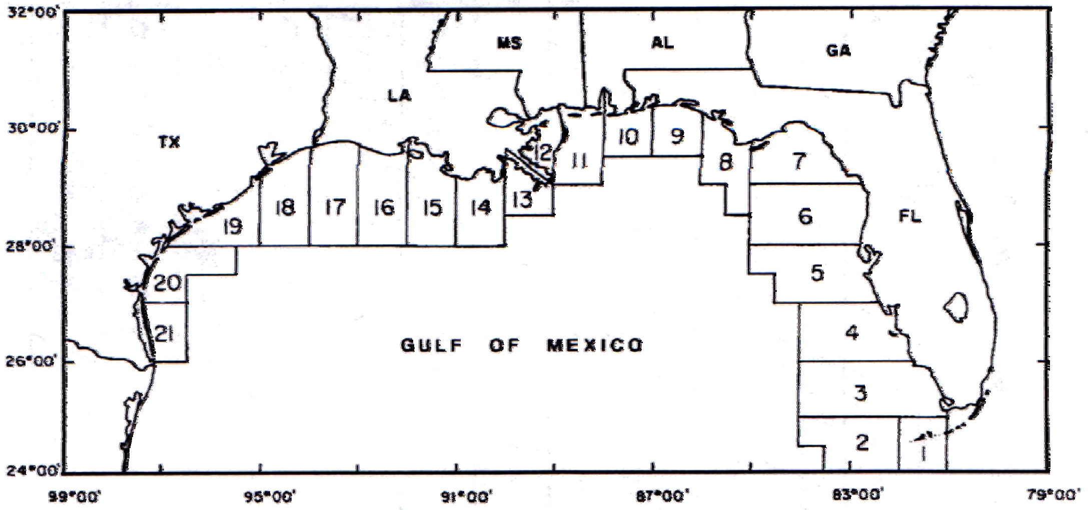


Figure 8. NMFS Statistical Sub-Areas for the Gulf of Mexico. (Data Source: TEWG, 2000).

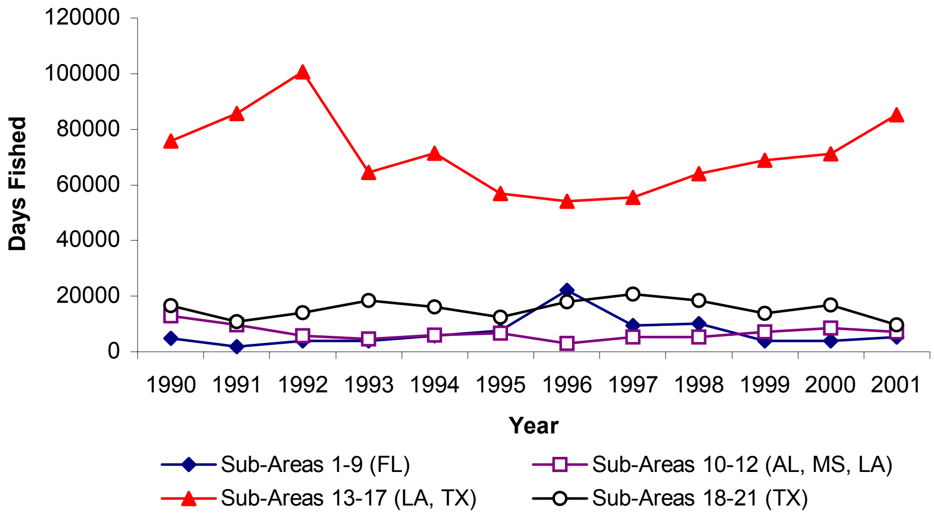


Figure 9. Annual commercial shrimping effort in offshore waters (<10 fm) of the Gulf of Mexico grouped by statistical sub-areas.

Godfrey and Cluse provided information on commercial fishery interactions contributing to North Carolina strandings via personal communications.

Data Analysis

Capture data were transformed into a measure of catch-per-unit-effort by dividing the number of ridleys captured in an entanglement net(s) by the product of number of hours the net(s) was deployed and cumulative length of the net(s). Monthly ridley CPUE values were then log transformed [Log (CPUE+1)] to approximate a normal distribution. One-way ANOVA was used to identify differences in ridley CPUE and size between years (across all sites combined and at Sabine and Calcasieu Passes), while the t-test was used to examine differences in these parameters between study areas (Sabine and Calcasieu Passes) and blocks of years (1993-1997 vs. 1998-2002). One-way ANOVA also was used to detect differences in ridley CPUE between sample months (April-October) and stations (1, 3, 4, 5) across all study sites combined. Post hoc comparisons were conducted using the Least Significant Difference (LSD) test. Trends in annual mean CPUE and size were analyzed via the null hypothesis that slope of the regression line was equal to zero ($\alpha = 0.05$), thus indicating no significant increasing or decreasing trend. The relationship between Kemp's ridley annual mean CPUE and number of hatchling released from Rancho Nuevo, Mexico (plotted with a 2-yr lag to account for the pelagic stage and estimated age of most turtles encountered) was analyzed via least squares linear regression analyses, with number of hatchlings transformed (-1/x) to linearize the plot of data points (Ott, 1993). Least squares linear regression also was utilized to assess the relationship between Kemp's ridley CPUE and hatch success at the nesting beach (plotted with a 2-yr lag to account for the pelagic stage) because fluctuations in this parameter may be representative of nesting female fecundity and nest conditions that produce viable hatchlings. The influence of number of hatchlings and hatch success at Rancho Nuevo on Kemp's ridley abundance in nearshore waters was examined via multiple regression. Pearson Correlation was used to assess the similarity between Kemp's ridley CPUE and stranding levels along the Texas coast during 1994-2002 (variables were not assessed as independent and dependent). Monthly and annual ridley abundance data also were examined for any autocorrelation between values that could confound regression analysis results. Neither monthly nor annual ridley CPUE was significantly autocorrelated (all Durbin-Watson test statistics ~ 2.0, over 72 lags for monthly CPUE and 9 lags for annual mean CPUE) (Ott, 1993). Mean values in this chapter are expressed as $\bar{x}$ ± standard error. All statistical analyses were conducted using an α level of 0.05 in Microsoft Excel and SPSS statistical software packages.

Results

Sea Turtle Captures – Northwestern Gulf

A total of 600 Kemp's ridley sea turtles was captured at sampling locations in the northwestern Gulf of Mexico during April-October 1993-2002. Sampling at Sabine Pass over the 10-year period produced 368 ridley captures, with the majority of these recorded prior to 1999 (343 prior vs. 25 after 1999). Wild, headstart, and wild-recaptured counterparts accounted for 90, 6 and 4%, respectively, of all ridley captures during

1993-1998. Thereafter, the proportion of wild caught ridleys increased to 96%, given an absence of recaptures. Although fewer ridleys were captured at Calcasieu Pass (232) during the 8 years this site was sampled (no sampling in 1996 and 1997), its post-1998 capture totals exceeded those for Sabine Pass (168 at Calcasieu vs. 25 at Sabine). The highest annual number of ridleys captured over the 10-year period was 105 at Calcasieu Pass in 1999. Percent contribution among wild, headstart and wild recaptured ridleys at Calcasieu Pass mirrored that at Sabine Pass, with a similar increase in the proportion of wild captures and a decrease in headstart and wild-recaptured conspecifics after 1998.

Netting Effort

Netting effort across all northwest Gulf study areas during 1993-2002 totaled 792.92 km-hours. Although total netting effort over the 10-year interval was greatest at Sabine Pass, Calcasieu Pass received highest annual effort after 1998.

CPUE

Monthly CPUE varied greatly within and across years (Table 2), with late spring and early summer yielding highest ridley capture rates (Fig. 10). Examination of monthly CPUE values across years and locations revealed significant variation between netting stations (ANOVA: $n = 190$, $F_{3, 186} = 4.803$, $p = 0.003$), with the western (stations 1 and 3) producing higher mean CPUE (t-test: $n = 217$, $t_{215} = 4.059$, $p < 0.001$) than did eastern counterparts. While this result suggests Kemp’s ridleys prefer conditions on the west or down current side of Sabine and Calcasieu Passes, bias

Table 2. Monthly Kemp's ridley CPUE at Sabine and Calcasieu Passes and annual statistics for all sites combined in the NW Gulf during 1993-2002.

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Sabine Pass										
April	1.01	2.36	0.31	0.15	0.63					
May	1.19	1.05	1.59	1.94	1.85	1.28	0.33	0.00	0.24	
June	0.50	1.99	0.62	0.22	1.50	0.11	0.00	0.00	0.00	0.31
July	1.28	0.22	1.07	0.99	1.09	0.00	0.11	0.00	0.12	0.57
August	1.20	2.73	0.48	0.12	0.43	0.11	0.18	0.00	0.16	
September	0.93	0.69	0.00	0.45	0.09					
October	0.13	0.26	0.00	0.00						
n	93	46	33	41	94	36	11	0	4	10
Mean	**0.89**	**1.33**	**0.58**	**0.55**	**0.93**	**0.38**	**0.15**	**0.00**	**0.13**	**0.44**
STD	0.43	1.03	0.58	0.70	0.67	0.61	0.14	0.00	0.10	0.18
STERR	0.16	0.39	0.22	0.26	0.25	0.23	0.05	0.00	0.04	0.07
CV	47.70	77.34	100.03	125.82	71.68	161.40	89.88	0.00	76.95	40.95
Calcasieu Pass										
May	0.79						1.60	0.56	1.20	0.82
June	0.47	0.00	0.00			0.71	1.07	0.42	0.25	0.18
July	0.16	1.93	0.50			0.45	0.52	0.31	0.58	0.73
August		0.48	1.25			0.14	1.63	0.32	0.13	
September	0.27	0.00					0.80			
October	0.00	0.00	0.00				0.67			
n	5	33	15			11	105	21	19	23
Mean	**0.34**	**0.48**	**0.44**			**0.44**	**1.05**	**0.40**	**0.54**	**0.58**
STD	0.31	0.84	0.59			0.29	0.48	0.12	0.48	0.34
STERR	0.14	0.37	0.26			0.13	0.21	0.05	0.21	0.15
CV	90.59	173.29	135.10			65.43	45.51	28.91	88.81	59.57
All Sites Combined										
n	98	79	48	41	94	47	116	21	23	33
Mean	**0.66**	**0.98**	**0.53**	**0.55**	**0.93**	**0.40**	**0.69**	**0.20**	**0.33**	**0.52**
STD	0.46	1.01	0.56	0.70	0.67	0.46	0.59	0.23	0.39	0.27
STERR	0.13	0.29	0.17	0.26	0.27	0.17	0.20	0.08	0.14	0.12
CV	69.95	103.47	105.75	125.82	71.68	114.66	85.12	113.41	116.24	51.63

Blank = Not Sampled

CPUE shown for Calcasieu 1999 and 2000 also includes that from Mermentau Pass, LA.
n = total number of individuals captured; STD = Standard Deviation; STERR = Standard Error; CV = Coefficient of Variation

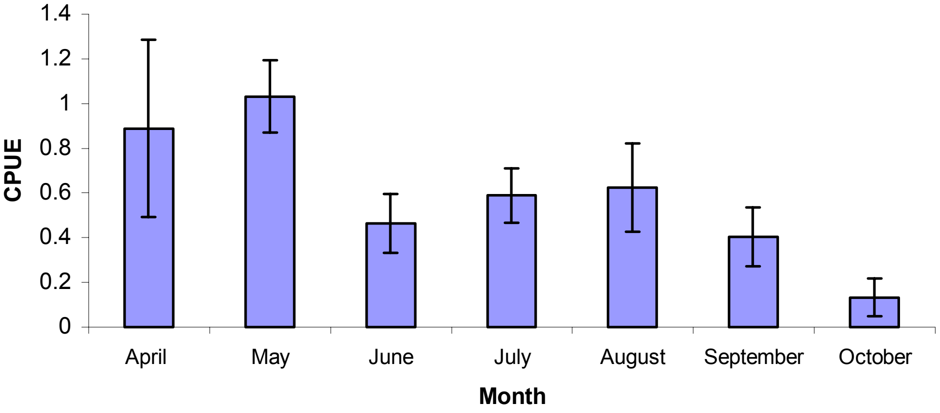
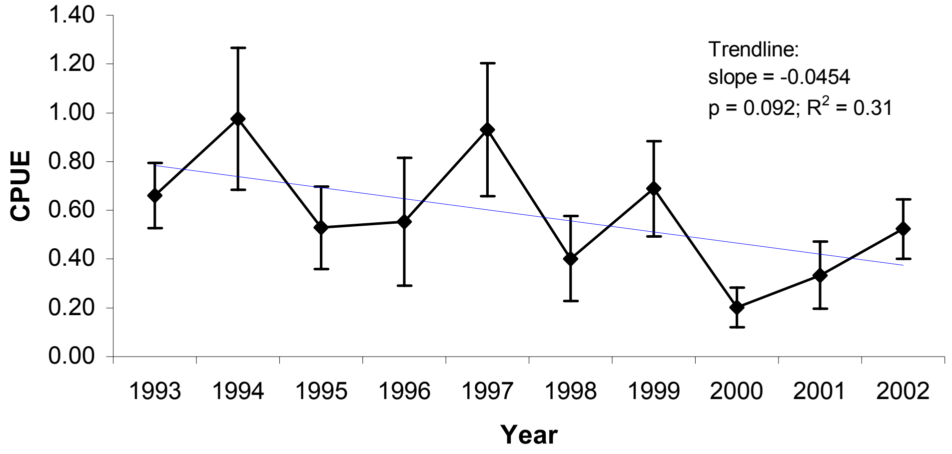


Figure 10. Mean *L. kempii* CPUE (w/ standard error bars) for months entanglement netting occurred at NW Gulf of Mexico sites during 1993-2002.

resulting from a disparate amount of netting effort concentrated on the west side of the jetties may underestimate this species' population trends east of the jetties. There were no significant differences in ridley CPUE detected between jetty and beach sampling stations ($n = 217$, $t_{215} = 1.801$, $p = 0.073$) across all study sites in the NW Gulf.

Mean annual Kemp's ridley CPUE across all sample sites and years combined is plotted in Figure 11a. CPUE peaks were observed in 1994 (0.98 ± 0.29/ km-hr, n = 12), 1997 (0.93 ± 0.67/ km-hr, n = 6), 1999 (0.69 ± 0.20/ km-hr, n = 10) and 2002 (0.52 ± 0.12/ km-hr, n = 5). Although CPUE statistics reflect a declining trend in ridley abundance, the slope of a line fitted to these data is not significantly different than 0 ($n = 10$, slope = -0.0454, $r^2 = 0.31$, $F_8 = 3.67$, $p = 0.092$), thus indicating a relatively constant abundance over the 10-year study. Variability in annual mean abundance was examined

a) All Study Areas Combined



b) Sabine and Calcasieu Passes

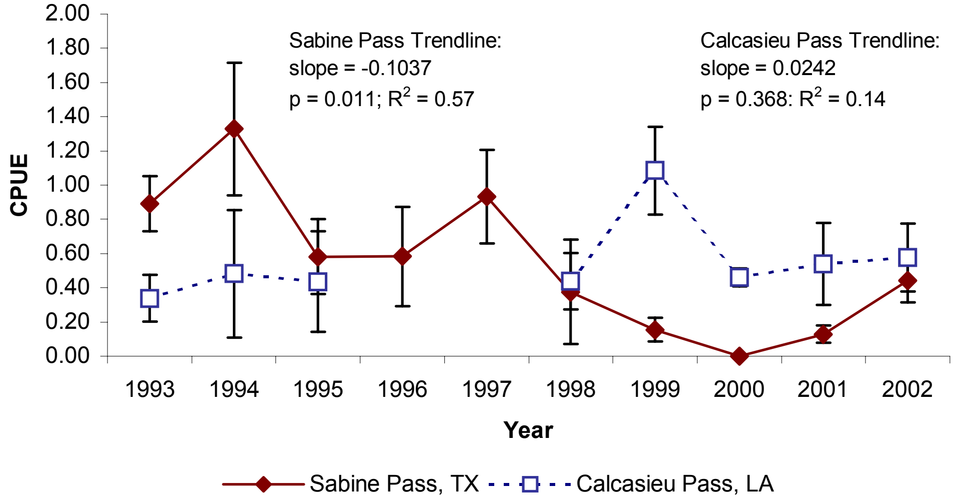


Figure 11. Annual mean Kemp's ridley CPUE (w/ standard error bars) for years entanglement netting occurred at all study areas combined (a) and Sabine and Calcasieu Passes (b) during 1993-2002.

via monthly CPUE values, with no significant difference detected between years (ANOVA: n = 86, $F_{9, 76} = 1.325$, p = 0.240).

Kemp's ridley CPUE at respective study areas across years is displayed in Figure 11b. Annual mean CPUE at Sabine Pass peaked in 1994 (1.33 ± 0.39/ km-hour, n = 7) and 1997 (0.93 ± 0.25/ km-hour, n= 6), then declined from 0.38 ± 0.23/ km-hour (n = 4) in 1998 to 0 in 2000 (Table 2). Thereafter, gradual increases in ridley catches through 2002 returned CPUE levels near the 1998 value. Overall, the slope of a trendline for ridley captures at Sabine Pass was negative and significantly different than zero (n = 10, slope = -0.1037, $r^2 = 0.57$, $F_8 = 10.76$, p = 0.011), indicating an annual decline in ridley CPUE at this site. This change in ridley abundance also was denoted by a significant difference in CPUE between years (ANOVA: n = 52, $F_{9, 42} = 2.909$, p = 0.009) and blocks of years, 1993-1997 versus 1998-2002 (t-test: n = 52, $t_{50} = 4.825$, $p < 0.001$).

Annual Kemp's ridley CPUE at Calcasieu Pass was relatively stable across years (n = 8, slope = 0.0263, $r^2 = 0.14$, $F_6 = 0.950$, p = 0.341, n = 10), with the exception of a major peak (1.05 ± 0.48/ km-hr, n = 6) in 1999 (Fig. 11b). Even with this peak, there was no significant difference in CPUE between years (ANOVA: n = 34, $F_{7,26} = 1.195$, p = 0.340) nor blocks of years, 1993-1997 vs. 1998-2002 (t-test: n = 34, $t_{32} = 1.812$, p = 0.079). A statistical comparison of monthly CPUE values from Sabine and Calcasieu Passes yielded significantly higher mean abundance at Sabine Pass prior to 1997 (n = 35, $t_{33} = 2.410$, p = 0.022) and the converse thereafter (n = 38, $t_{36} = -4.196$, $p < 0.001$).

Size Composition

Straight carapace length (cm) of Kemp's ridleys captured in nearshore Gulf waters during the 10-year study ranged from 19.5 to 66.3 cm. Average size peaked in 1995 (39.9 ± 1.42 cm, n = 48) and 2000 (39.7 ± 1.86 cm, n = 21), but annual means did not exhibit a significant trend (n = 10, slope = 0.0833, $r^2 = 0.01$, $F_8 = 0.090$, p = 0.766) (Fig. 12). A significant size difference was detected across years (ANOVA: n = 600, $F_{9, 540} = 3.217$, p = 0.001), but all annual mean size values fell between 30 and 40 cm SCL. Ridley size at Sabine Pass ranged from 19.5 to 64.0 cm SCL, while that for Calcasieu Pass counterparts varied between 22.4 and 66.3 cm SCL. There appears to be a reduction in the size range of ridleys captured at Sabine Pass after 1998, with an absence of individuals larger than 55.0 cm (Fig. 12). Ridleys were significantly larger ridleys at Calcasieu Pass ($\bar{x}$ = 37.2 ± 0.62 cm SCL, n = 234) than at Sabine Pass ($\bar{x}$ = 35.0 ± 0.47 cm, n = 368) (t-test: n = 600, $t_{598} = -2.869$, p = 0.004). Seventy-seven percent of all ridleys captured during the study were between 20 and 40 cm SCL, with larger individuals uncommon (40-60 cm: ~ 20%; > 60 cm: 2%). Calcasieu Pass yielded a higher percentage of ridleys greater than 40 cm (28%) when compared to conspecifics from Sabine Pass.

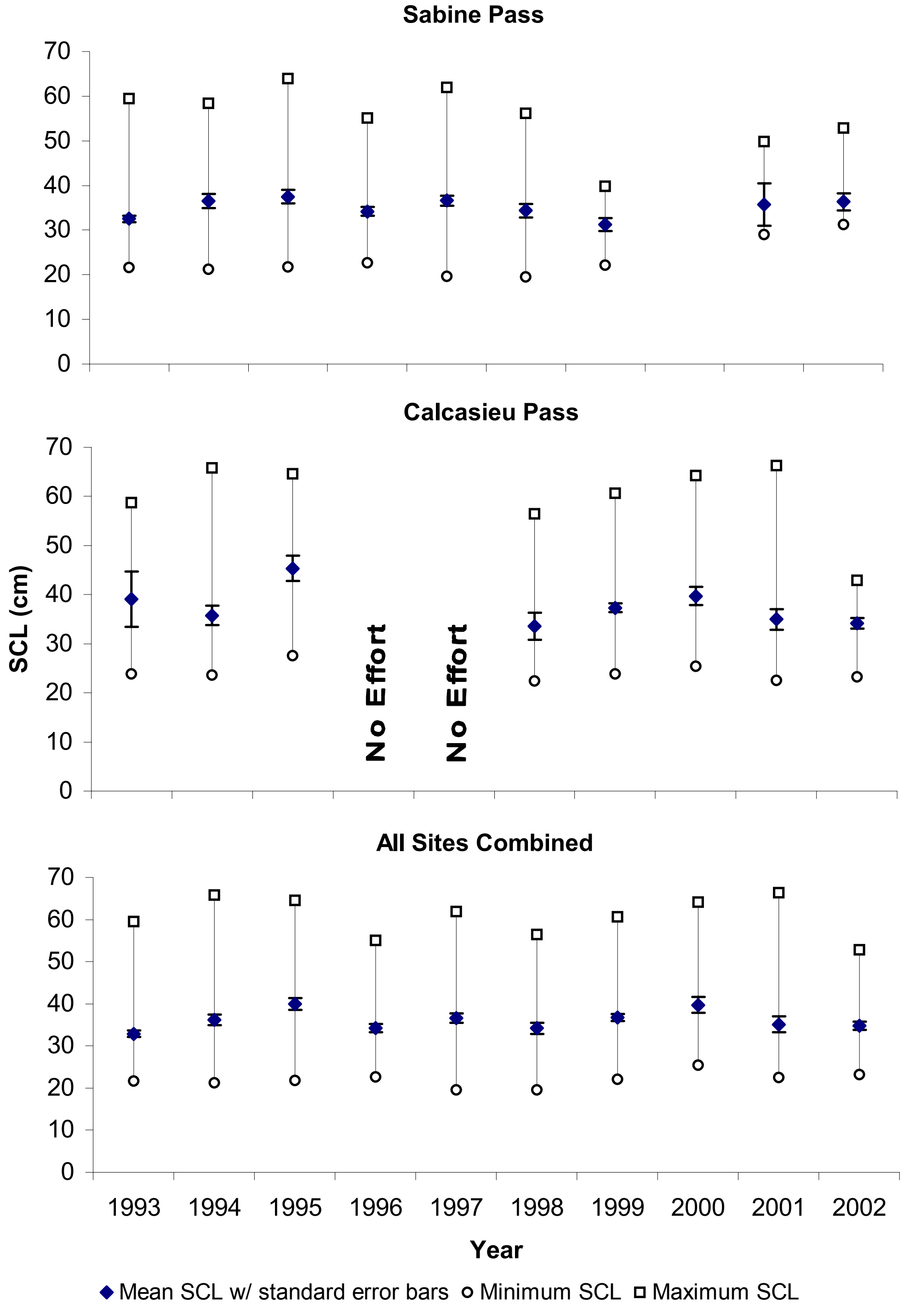


Figure 12. Annual mean size of Kemp's ridleys captured at Sabine and Calcasieu Passes and all sites combined in the NW Gulf during 1993-2002.

Nesting Dynamics

The number of Kemp's ridley nests, eggs and hatchlings released from the Rancho Nuevo, Mexico nesting beach since the late 1980's has increased exponentially (Fig. 13). For example, the 405,544 hatchlings released in 2002 represent nearly a 10-fold increase over that for 1987 (44,634). Number of nesting females in 2002, based on 2.5 nests/female/season (TEWG, 2000), was estimated to be 2,574. This is approximately 8 times greater than the estimated number of nesters in the 1980's, but about 7,400 females less than the recovery goal of 10,000 adult females in a season (Table 3). Hatch success was stable during 1992-2002 ($n = 11$, slope = -1.0727, $r^2 = 0.31$, $F_9 = 4.04$, $p = 0.075$), but slightly higher (50-80%) than that in the late 1970s/early 1980s (40-70%). Estimated annual clutch size (number of eggs per nest) has shown a declining trend since 1978, with the average going from 100-103 eggs/nest in the early 1980s to 90-95 eggs/nest in the late 1990s (Table 3). Patterns in estimated annual clutch size since 1992 was very similar to overall ridley CPUE in northwestern Gulf developmental habitats, with both exhibiting peaks in 1994, 1997 and 1999.

Relationship of Nearshore Kemp's Ridley Abundance to Nesting Productivity

As number of hatchlings released from the Rancho Nuevo nesting beach increased from 1992 to 2002 ($n = 11$, slope = 33075.9, $r^2 = 0.84$, $F_9 = 45.81$, $p < 0.001$), ridley CPUE at study areas in NW Gulf either remained stable or declined slightly (Fig. 14). A comparison between transformed (-1/x) annual number of hatchlings released

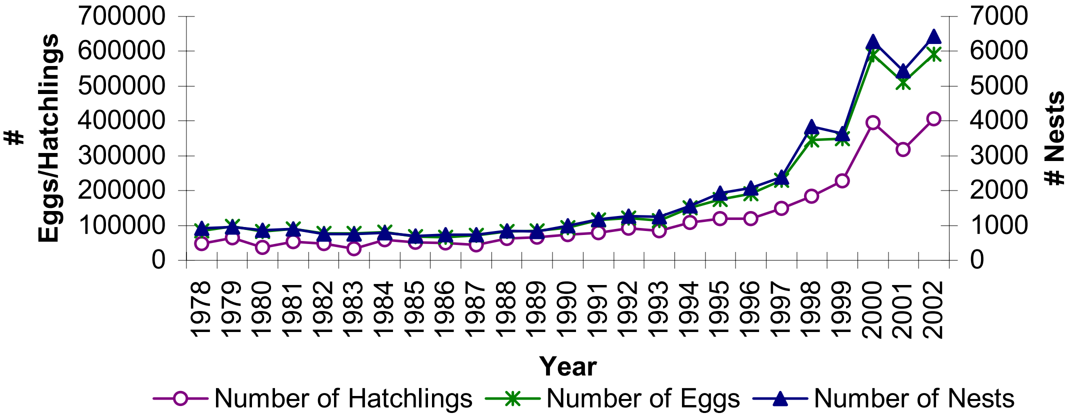


Figure 13. Annual number of Kemp's ridley nests, eggs and hatchlings at the Rancho Nuevo, Tamaulipas, Mexico nesting beach during 1978-2002.

Table 3. Nesting statistics (number of nesting females, average clutch size, and hatch success) for Kemp's ridleys at Rancho Nuevo and adjacent beaches estimated from number of nests, eggs, hatchlings during 1978-2002. (Source: KRWG reports)

Year	Nesting Females	Clutch Size	Hatch Success
1978	370	92.2	56
1979	382	102.9	65
1980	347	94.9	45
1981	359	100.2	59
1982	300	103.7	62
1983	298	103.8	43
1984	319	101.3	72
1985	281	96.3	75
1986	298	87.8	75
1987	295	97.9	62
1988	337	98.8	75
1989	331	102.1	79
1990	397	94.7	79
1991	471	97.6	69
1992	510	94.9	76
1993	496	91.6	74
1994	625	96.1	72
1995	772	90.4	68
1996	832	92.3	62
1997	955	96.3	65
1998	1538	90.1	53
1999	1456	95.9	65
2000	2511	94.0	67
2001	2177	93.7	62
2002	2574	92.0	68

Nesting females = # of nests divided by 2.5 nests/female/season
Clutch size = # eggs divided by # of nests
Hatch success = # hatchlings/ # eggs

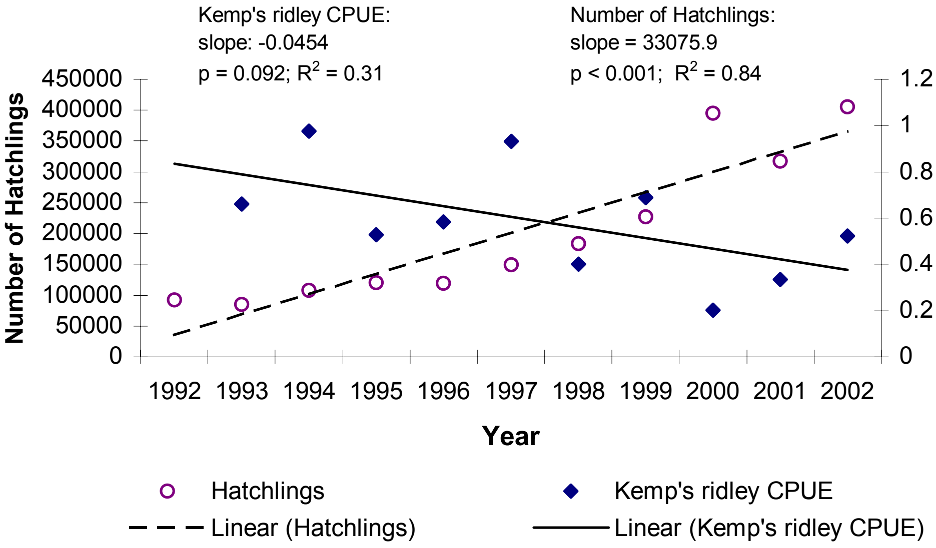


Figure 14. Trends in annual number of Kemp's ridley hatchlings released from the Rancho Nuevo, Mexico nesting beach and ridley CPUE in the NW Gulf during 1992-2002.

from Rancho Nuevo and Kemp's ridley CPUE in developmental habitats (Fig. 15; plotted with a 2-year lag to account for the pelagic stage and estimated age of nearshore ridleys, 30-35 cm SCL) yielded a weak, statistically non-significant negative relationship (Linear regression: $n = 9$, $r^2 = 0.23$, $F_{1,7} = 2.089$, $p = 0.192$). Approximately 50% of the variability in coastal ridley abundance patterns was explained by hatch success 2 years before ($n = 9$, $r^2 = 0.49$, $F_{1,7} = 6.740$, $p = 0.036$) (Fig. 16). Multiple regression analysis revealed that there was no statistically significant relationship between Kemp's ridley CPUE and the combination of both number of hatchlings (-1/x transformed) and hatch success ($n = 10$, $r^2 = 0.50$, $F_{2,6} = 2.941$, $p = 0.129$).

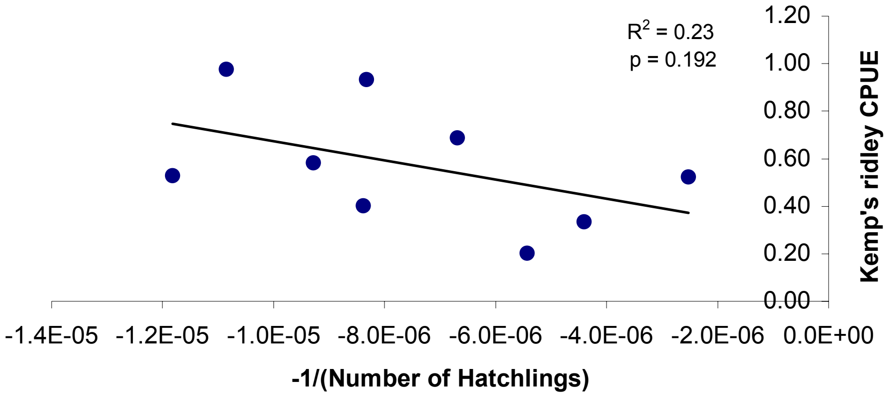


Figure 15. Annual mean Kemp's ridley CPUE in the NW Gulf versus number of hatchlings (transformed -1/x) released from Rancho Nuevo, Mexico, plotted with a 2-year lag to account for the pelagic stage.

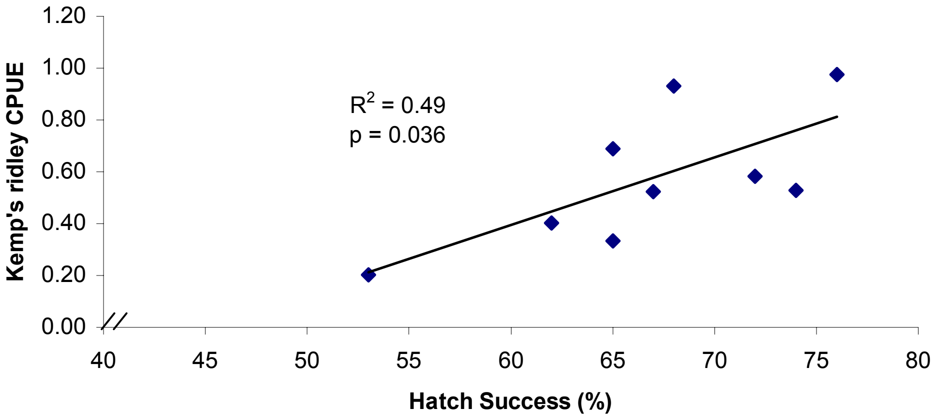


Figure 16. Annual Kemp's ridley CPUE in the NW Gulf versus hatch success at the Rancho Nuevo, Mexico nesting beach (1992-2002), plotted with a 2 year lag to account for the pelagic stage.

Strandings Data

Sea turtle stranding statistics along the Texas coast for all species during 1994-2002 are provided in Appendix B (only total number of stranded turtles was available for this assessment). Of all species contributing to these strandings, only the Kemp's ridley has exhibited a slightly significant decreasing trend ($n = 9$, slope = -13.967, $r^2 = 0.55$, $F_7 = 8.61$, $p = 0.041$). Also, the percentage of ridleys contributing to annual stranding totals decreased slightly from nearly 50% in 1994 to about 25-30% during 1999-2002. The pattern in number of Texas ridley strandings since 1994 is extremely similar to that of annual mean ridley CPUE at Sabine Pass (Pearson correlation: $n = 9$, $r = 0.927$, $p < 0.001$) (Fig. 17).

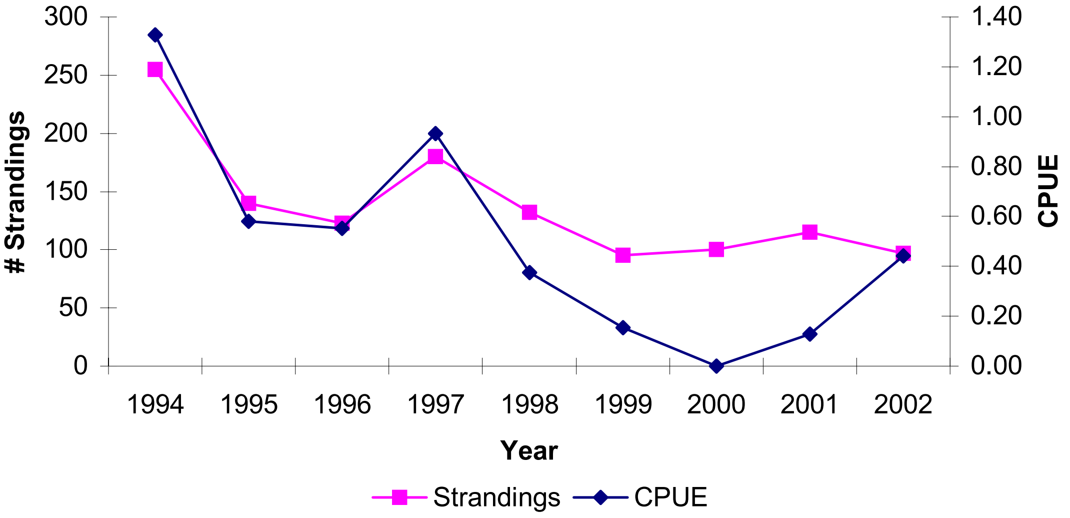


Figure 17. Annual Kemp's ridley strandings along the Texas coast and CPUE at Sabine Pass, TX during 1994-2002.

Kemp's ridley strandings in Florida have exhibited a significant increasing trend since 1987 (A. Foley, *personal communication,* July 2003). An average of 40 ridleys annually stranded in Florida during 1987-1994 amounting to about 5% of the state's overall total sea turtle strandings. Subsequently, these strandings increased to about 100 per year or 9% of the overall total sea turtle strandings. Also, about 3.6 times more ridleys have stranded along Florida's Gulf coast compared to along its Atlantic coast. Stranded ridleys in Florida exhibited a wide range of sizes, with a mean curved carapace length (CCL) of 43.4 cm (≅ 41.8 cm SCL based on regression equation, CCL = 1.0345 SCL + 0.1162, reported by Schmid [1998]).

Kemp's ridley strandings in North Carolina (1993-2002) increased sharply from 42 in 1997 to 113 in 1999, but declined thereafter, to return to 40 by 2002. Peak North Carolina strandings in 1999 was largely attributed to a significant increase in the gill-net effort for monkfish, a fishery that was consequently closed after spring 2000 (Wendy Cluse, *personal communication,* July 2003). Thus, elevated North Carolina strandings were mostly likely not representative of an increased ridley presence in this region. Size distribution of stranded ridleys in North Carolina across years exhibited an increasing trend in percentage of constituents greater than 40 cm CCL, resulting in a more equitable contribution of small and large immature ridleys by 2002.

Discussion

Characterization of Kemp's ridley long-term abundance and distribution in nearshore waters of the northwestern Gulf reveals a relatively consistent, 2-3 year cycle in ridley CPUE across study areas. Elevated capture rates in the late spring/early summer are consistent with seasonal occurrence patterns reported in other studies from the Gulf (Ogren, 1989; Rudloe *et al.*, 1991; Manzella and Williams, 1992; Schmid, 1998; Landry and Costa, 1999). However, sampling in this study was restricted to months in which ridleys were more likely to be present and, as such, did not allow for comparison across all seasons. Inconsistent sampling of certain months (e.g. April, September and October) during this 10-year survey also may have biased these results. Similarly, higher abundance of ridleys observed at netting stations west of the jetties is confounded by greater sampling effort directed at these locations, creating a larger sample size, over that for eastern counterparts.

Juveniles and subadults (20-40 cm SCL) dominated the nearshore ridley assemblage (77%), although large immatures and adults also were encountered. Age estimates of wild ridleys using skeletochronological data and the von Bertalanffy growth equation suggest the majority of ridleys in this size range are 2-3 years old (Appendix A) (Zug *et al.*, 1997; TEWG, 2000). These size/age observations are consistent with ridleys that have recently transitioned to nearshore feeding grounds from the pelagic stage (Ogren, 1989; Manzella and Williams, 1992; Landry and Costa, 1999). The slightly smaller size of ridleys at Sabine Pass (35.0 cm mean SCL vs. 37.2 cm mean SCL at Calcasieu) may indicate that it was the first nearshore location encountered during recruitment to neritic habitat from the pelagic stage, yet this difference in size could simply be due to sampling variability. It also may imply that Kemp's ridleys move east toward Calcasieu Pass over time. Alternatively, the higher frequency of large turtles at

Calcasieu Pass may be due to adult females (> 60 cm) using foraging habitat in coastal waters off the Louisiana coast for reproductive conditioning (Hildebrand, 1982; Renaud *et al.*, 1996). Kemp's ridleys 40-60 cm SCL from the eastern Gulf also may be moving westward toward the nesting beach at Rancho Nuevo.

Sabine and Calcasieu Passes are historical index habitats of ridley occurrence (Landry and Costa, 1999), and, thus, it is not surprising that larger individuals are occasionally captured. However, the size distribution of Kemp's ridleys at Sabine Pass shifted over time to include smaller and fewer turtles overall. This pattern might signify a change in the attractiveness of this location as a foraging ground for adults as well as juveniles. Radio-tracking studies on juvenile ridleys near Sabine and Calcasieu Passes by Renaud *et al.* (1995) indicated these young turtles exhibit strong site fidelity to jettied tidal passes. Yet, decreased frequency of recaptures at both sites after 1998 may suggest reduced attractiveness (i.e., foraging quality) that could cause turtles to disperse in search of better conditions (i.e., prey). The observed shift in ridley abundance from Sabine Pass to Calcasieu Pass after 1997 may be related to changes in foraging success or other biotic and abiotic factors that are examined in later chapters, or factors that were not examined in this study.

Increases in number of nests, eggs and hatchlings released from Rancho Nuevo since the early 1990s probably resulted from: 1) nesting beach protection and better husbandry of hatchlings; and 2) increased survivorship of coastal immature ridleys due to the use of TEDs in shrimp trawls during the past 7 years that has resulted in an increased number of nesting females (TEWG, 2000). Fluctuations in hatch success and average estimated clutch size are likely related to age/size of nesters, reproductive conditioning of females, and a variable re-migration interval. Tag returns, indicate that 20% of adult females return to nest every year, 60% every 2 years, 15% every 3 years, and 5% every 4 years (TEWG, 2000). Older, more fecund females may only nest every 2 or 3 years, producing peaks in hatchling production (Shine, 1980; Frazer and Richardson, 1986; Márquez *et al.*, 1989; Olson and Shine, 1996; Madsen and Shine, 1999; Broderick *et al.*, 2001; Broderick *et al.*, 2003). Decreased average ridley clutch size since 1978 may be due to an influx of neophyte nesters in recent years (Márquez, 1994), presumably resulting from increased survivorship of juveniles and sub-adults. There is a known relationship between clutch size and turtle body size (Frazer and Richardson, 1986). Thus, with more young females nesting for the first time, and as the population grows, it is expected that average clutch size in the ridley nesting population would decrease.

Kemp’s ridley abundance patterns in nearshore nursery habitat were hypothesized to correlate with nesting activity. Based on the increasing trend in number of hatchlings released from Rancho Nuevo since the early 1990s, a corresponding increase in juvenile abundance at nearshore sampling locations could be expected with some time lag to allow for the pelagic stage. However, juvenile ridley CPUE at netting sites remained stable or decreased slightly while number of hatchlings produced at Rancho Nuevo continued to increase. Examination of nearshore ridley abundance and nesting success, with a 2-year lag to account for the “Lost Years”, yielded no significant relationship between coastal ridley CPUE and number of hatchlings. It is possible that any similarities in cycles of nearshore CPUE and hatchling numbers may be masked by the exponentially increasing trend presently observed in the latter. Conversely, number of hatchlings produced annually still may not be sufficient to produce significant increases in nearshore recruitment, given the high mortality of post-hatchlings. Yet, this does not explain the higher CPUE values during years when nesting productivity was much lower (i.e., early 1990s). Another possibility is that these juvenile ridleys are recruiting to coastal locations outside of the NW Gulf study areas.

The significant relationship between ridley CPUE and hatch success (with a 2-year lag) also may be linked to greater hatchling or post-hatchling cohort survival, producing 2-3 year pulses in nearshore recruitment. The similarity in cycles present at the nesting beach and in nearshore waters may reflect the variability in nesting female fecundity and nest conditions that influence the number of hatchlings that leave the nesting beach, the survivors of which are available to recruit to coastal waters. Yet, interpretation of these results also is complicated by the presence of several year classes in developmental habitat at a given time and variable duration of the pelagic stage (1-4 years) and nesting female remigration interval (1-4 years) (TEWG, 2000).

Assuming mortality rates did not drastically increase during the monitoring period, one would expect an increase in juvenile ridley abundance somewhere. The comparison of abundance at Sabine and Calcasieu Passes and reproductive output at Rancho Nuevo is limited. A survey of ridley strandings across multiple regions may provide the data necessary to adequately address this issue, so long as factors affecting stranding rates are taken into consideration. A recent study by Lewison *et al.* (2003)

reviewed stranding records for loggerhead and Kemp's ridley sea turtles in the western Gulf of Mexico from 1986-2000. The authors reported the highest number of Kemp's ridley strandings in 1994, despite more stringent TED regulations, and significantly higher strandings (22%) during 1994-2000 (w/mandatory TED implementation) compared to those in 1986-1993 (w/voluntary TED implementation). They also determined that elevated stranding rates from 1994-2000 were evidence of in-water population increases, because shrimping effort and TED compliance has remained relatively constant. However, Lewison et al. (1994) did not address the slight declining trend in Texas ridley strandings since 1994. A similar declining trend also was observed with in-water abundance patterns at Sabine Pass, and may indicate a shift in distribution of turtles to other regions along the Gulf and Atlantic coast. However, elevated North Carolina strandings in 1999 were largely attributed to a significant increase in the gill-net effort for monkfish and mostly likely not representative of an increased ridley presence in this region.

Ridley strandings in Florida have increased since 1995, despite relatively constant shrimping effort, (A. Foley, *personal communication,* July 2003) and thus may be representative of increased ridley presence in the eastern Gulf. This idea has not been supported with data from in-water abundance surveys from Florida (J. Schmid and W. Witzell, *personal communications,* June-July 2003). However, this lack of corroboration may be due to varying survey methods and short duration of sampling (when compared with this study) that did not provide adequate CPUE for meaningful comparison across years. In any case, if ridley strandings are representative of larger scale abundance and distribution patterns, then there is evidence to suggest a change in population dynamics of *L. kempii* in the Gulf of Mexico. Many factors could influence distribution of turtles in the Gulf, including changes in circulation patterns that transport post-pelagic turtles to the neritic zone, or changes in the quality of nearshore foraging habitat at NW Gulf study sites.

Summary

Analyses conducted in this chapter provide valuable data on Kemp's ridley long-term in-water abundance and distribution, particularly for juveniles in developmental habitat of the northwestern Gulf. The 2-3 year cycle in ridley CPUE observed across all study sites was related to hatch success at Rancho Nuevo. Patterns in this nesting parameter depend on adult female fecundity and the variable remigration interval. Yet, there was no significant relationship between ridley CPUE in nearshore waters of the NW Gulf and the increasing number of hatchlings released from the nesting beach (with a 2-year lag) in recent years. In fact, ridley abundance at Sabine Pass exhibited a declining trend over the 10-year study period. Reduced occurrence at this historic index habitat may be related to changes in foraging habitat quality. A decline in Texas ridley strandings since 1994 and concurrent increases in ridley strandings along the Florida coast suggest a shift in distribution may be occurring, with greater number of ridleys from elevated nesting activity recruiting to the eastern Gulf. However, use of strandings as a measure of long-term abundance and distribution is limited by often unpredictable and non-uniform factors contributing to stranding events. Future research should utilize consistent and un-biased in-water surveys at additional locations, especially along the

Gulf coast, to adequately assess Kemp’s ridley occurrence in nearshore waters. Although a clear relationship between coastal ridley abundance and nesting dynamics was difficult to discern, population monitoring aimed at all life history stages is an essential component in the management of this endangered species.

CHAPTER III

ABIOTIC FACTORS

Introduction

Abiotic factors are an integral component in Kemp's ridley sea turtle ecology due to their impact on distribution, habitat use and health. Because ridleys utilize varying habitats during different life history stages (Fig. 1), they are subject to both aquatic and terrestrial environments. Additionally, environmental conditions are rarely constant over time and may fluctuate on a diurnal, seasonal, annual, or multi-annual basis. Thus, Kemp's ridley abundance and distribution in nearshore developmental habitat may be significantly influenced by one or more environmental parameters on differing spatial and temporal scales.

The hypothesized influence of both pelagic and nearshore abiotic factors on ridley occurrence is expressed in the conceptual model described in Chapter I (Fig. 2; Table 1), with the specific portion of the model that pertains to these influences isolated in Fig.18. Although the influence of Gulf circulation patterns on ridley dispersal from pelagic to coastal benthic habitat is acknowledged in the conceptual model (Carr, 1980, 1987; Carr and Meylan, 1987; Collard and Ogren, 1990), difficulty in quantifying constituent variables prevents an in-depth assessment of their effect. The primary focus of this chapter is to characterize nearshore factors (i.e. monthly and annual trends in water temperature, salinity and visbility) at ridley population monitoring locations, and test for correlations between these environmental parameters and Kemp's ridley CPUE.

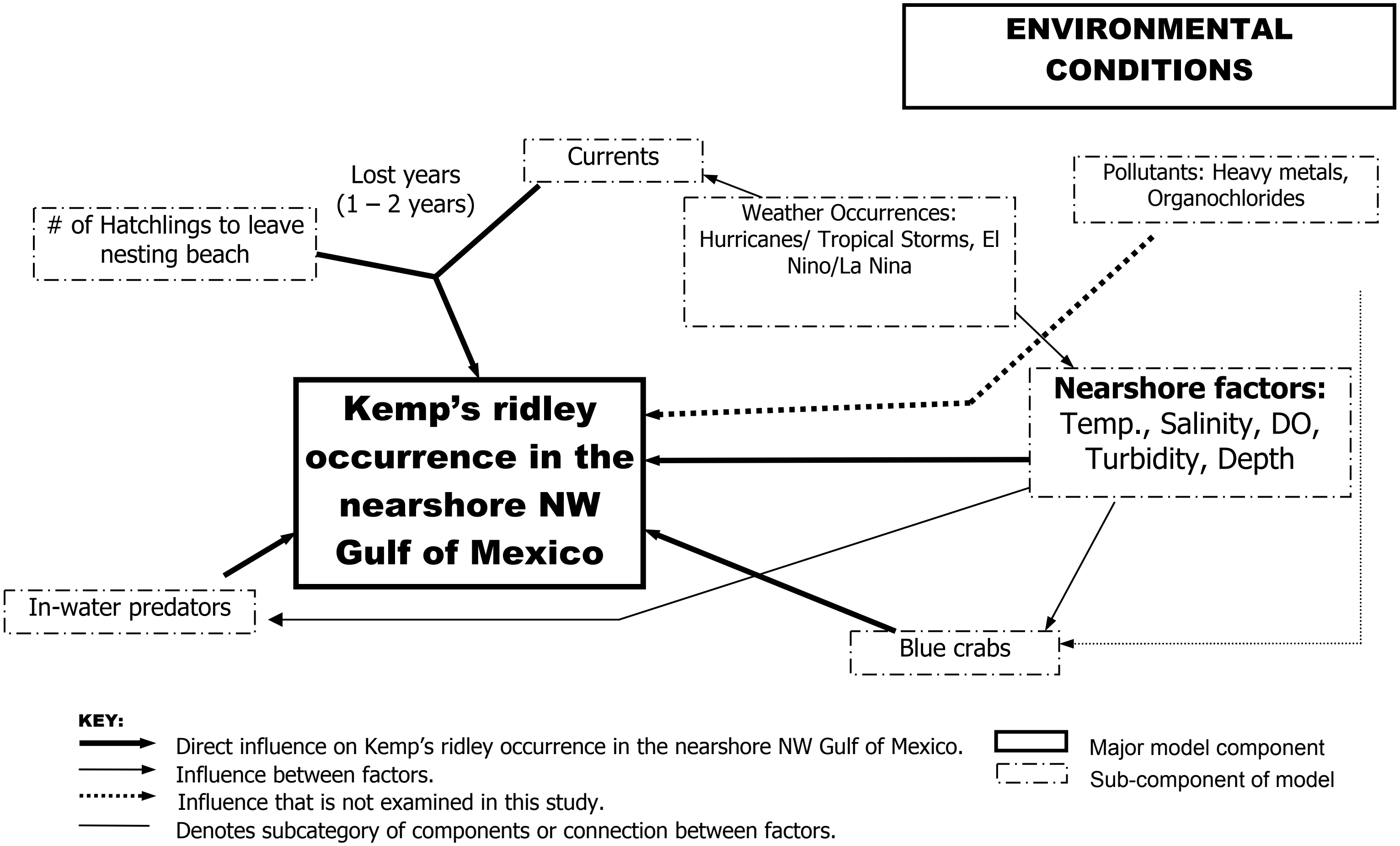


Figure 18. Portion of conceptual model detailing the hypothesized influence of environmental conditions on Kemp's ridley occurrence in nearshore waters of the NW Gulf of Mexico.

Nearshore Factors

Once Kemp's ridleys are 1-4 year(s) old, they move into shallow coastal waters to forage on benthic prey. Several abiotic factors, including weather, sea state, tidal flux, water depth, water temperature, salinity, and turbidity, may affect abundance and distribution in nearshore habitat; however, this study only examines the latter three. Seasonal occurrence and movements of ectotherms like the Kemp's ridley have typically been linked to water temperature (Lazell, 1980; Lutcavage and Musick, 1985; Henwood and Ogren, 1987; Byles, 1989; Ogren, 1989; Burke *et al.*, 1994; Landry and Costa, 1999). Juvenile ridleys in the Gulf of Mexico enter shallow coastal waters and bays during summer months, but they move offshore from October-December presumably to avoid water temperatures less than 20°C (Ogren, 1989; Landry and Costa, 1999). Seasonal ridley occurrence also is observed along the US east coast, but with offshore and southward movement in response to falling temperatures (Lazell, 1980; Lutcavage and Musick, 1985; Henwood and Ogren, 1987; Byles, 1989; Ogren, 1989; Burke *et al.*, 1994). However, rapid temperature drops that turtles cannot escape may result in turtles being stunned from the cold (Lazell, 1980; Meylan and Sadove, 1986). Moon *et al.* (1997) investigated behavioral and physiological responses of ridleys to decreasing water temperatures in a controlled environment and found those exposed to conditions less than 20°C initially became very agitated and active. Increased ridley activity with the onset of temperatures below 20°C may be indicative of movement to avoid thermal stress. The ridleys in the aforementioned experiment also ceased feeding and became semi-dormant as temperatures gradually fell below 15°C, but did not exhibit cold stunning behavior. This cessation in feeding and semi-dormant state might be a prelude to a ridley hibernation response to cold temperatures. Reduced occurrence of ridleys in shallow waters of the northern Gulf during winter months as well as reports of mud-covered turtles captured in colder waters by fishermen also may indicate that ridleys hibernate by partially burying in the bottom sediment (Ogren and McVey, 1982; Moon *et al.*, 1997; Landry and Costa, 1999). Summer water temperatures in the nearshore Gulf normally reach 32°C, but there is no evidence of adverse effects of warm water on Kemp's rideys.

Juvenile ridleys are often found in shallow bays and coastal waters near estuaries foraging on crabs. Estuaries are semi-enclosed bodies of water where rivers meet the ocean, and thus are areas of fresh and saltwater mixing. Positive estuaries, such as Sabine Lake (Sabine-Neches estuary), typically exhibit an increasing salinity gradient from their upper to lower reaches (Britton and Morton, 1989). However, this gradient may be highly dynamic in relation to periods of flood and drought. Annual rainfall for study locations in the NW Gulf averages 127 cm per year (Owenby *et al.*, 1992), but there have been periods of drought in this region, particularly in 1996 and 1998-2000 (Texas Water Development Board [TWDB], 1999; Wilson, 2001, National Climatic Data Center [NCDC], 2003). Freshwater inflow for the Sabine-Neches estuary is the highest of all major Texas bay systems, with the optimal ecological inflow determined to be 11.8 million cubic km/year (Texas Natural Resources Conservation Commission [TNRCC], 2002). As such, ridleys may be exposed to wide fluctuations in salinity, and presumably have mechanisms to regulate water loss and salt accumulation (Lutz, 1997).

The salt gland of sea turtles is a primary osmoregulatory organ that allows them to excrete excess salts through their tear ducts (Ortiz *et al.*, 2000). Ortiz *et al.* (2000) examined effects of acute freshwater exposure on osmotic responses in Kemp's ridleys. Exposure of these turtles to freshwater for 4 days resulted in increased water consumption and decreased ionic and osmotic concentrations. Although prolonged exposure to freshwater without salt supplements may cause decreased osmoregulatory capacity, there were no deleterious effects exhibited by ridleys during the 4-day experiment. Thus, ridleys inhabiting estuaries for short-term foraging purposes are probably euryhalinic in their ability to tolerate salinity fluctuations. However, salinity fluctuations may impact prey availability (i.e. blue crab abundance and distribution), thereby indirectly affecting Kemp's ridley use of nearshore waters.

Shallow coastal and estuarine waters of the NW Gulf are often quite turbid due to the high silt and sediment discharge of the Mississippi River carried westward by longshore currents. This results in nearshore water visibilities less than 1.0 m. Although there is a lack of information on underwater sight capabilities of the Kemp's ridley, Bartol *et al.* (2002) used electrophyisology and visually evoked potentials to measure the in-water visual acuity of juvenile loggerhead sea turtles and found that levels were comparable to those of other benthic, shallow water species. This result, combined with retinal morphology and behavioral studies, suggests that loggerhead sea turtles are capable of discerning horseshoe crabs and blue crabs, as well as large predators (Bartol and Musick, 2001; Bartol *et al.*, 2002). However, that experiment was conducted using filtered seawater in a goggle over the loggerhead's eye, and, as a result, may not be representative of sea turtle vision under natural conditions. As such, it is not known how turbidity may affect visual acuity in sea turtles, but the scattering of light by suspended particles might reduce Kemp's ridley's reactive distance when visually locating prey and avoiding predators or underwater obstacles (e.g., entanglement nets) (Benfield and Minello, 1996).

Based on results of previous studies, it is hypothesized that Kemp's ridley abundance will be positively correlated with water temperature, with greatest occurrence coinciding at temperatures ≥ 20ºC. Although ridleys are adapted to the marine environment, they have exhibited euryhalinic tolerance to salinity extremes during experimental conditions and thus will not be significantly affected by this factor. High turbidity may reduce the ability of ridleys to visually locate and avoid entanglement nets, consequently producing greater ridley CPUE. Conversely, ridleys may be deterred from inhabiting highly turbid areas due to reduced visual acuity, and this may result in lower CPUE at study areas.

Materials and Methods

Data Collection

Study areas and Kemp's ridley entanglement netting procedures are as described in Chapter II. Water temperature (°C), salinity (ppt), and visibility (m) recordings from Sabine and Calcasieu Passes (April to October 1993-2001) were measured at each netting site in the morning immediately after entanglement nets were deployed (0800 h), early afternoon (1300 h), and late afternoon at net retrieval (1700 h). Temperature and salinity were measured using a YSI datasonde, and visibility was measured with a Secchi disk. Rainfall data and freshwater inflow to the Sabine-Neches estuary also were gathered from various sources to assess their possible effect on salinity. Monthly and annual precipitation (cm) for the upper Texas coast (Matagorda Bay-Sabine Pass) and southwestern Louisiana were obtained online from the Western Regional Climate Center (http://www.wrcc-dri.edu). Freshwater inflow data (cubic km/month or year) for only the Sabine-Neches estuary 1992-1999 were acquired from the Texas Water Development Board website (http://hyper20.twdb.state.tx.us/data/bays_estuaries/hydrology/sabinesum.txt).

Data Analysis

One-way ANOVA was used to identify differences in the aforementioned environmental parameters between years (across all sites combined and at Sabine and Calcasieu Passes), while the t-test was used to examine differences in these parameters between study areas (Sabine and Calcasieu Passes). One-way ANOVA also was used to detect differences in these abiotic factors between sample months (April-October) for all study sites combined. Post hoc comparisons were conducted using the Least Significant Difference (LSD) test. Trends in annual mean values for these parameters were analyzed in the same manner as those for Kemp's ridley CPUE, with a null hypothesis that the slope of the regression line is equal to zero. Monthly and annual mean values were used in least squares linear regression to assess the relationship between Kemp's ridley abundance and each abiotic factor. Similarities in rainfall patterns between the upper

Texas and southwestern Louisiana coasts were examined via Pearson correlation (variables were not assessed as independent and dependent). Monthly mean values of water temperature and salinity in this chapter are expressed as $\bar{x}$ ± standard deviation, while monthly mean visibility and annual mean values for all parameters are expressed as $\bar{x}$ ± standard error. All statistical analyses were conducted using an α level of 0.05 in Microsoft Excel and SPSS statistical software packages.

Results

There were no significant trends in annual mean values for any environmental parameter measured across all locations in the NW Gulf (Table 4), but fluctuations in all factors produced significant differences between years (Table 5). Water temperature across all sites and years ranged from 14.5 to 35°C. Monthly mean temperature at Sabine and Calcsieu Passes exhibited an increasing trend from April to August within each sample year (ANOVA: $n = 1239$, $F_{6,\,1232} = 301.376$, $p < 0.001$) (Fig. 19). However, annual mean temperature was fairly stable across years at both study areas (Table 4). Somewhat lower water temperatures were observed at Sabine Pass ($\bar{x} = 28.3 \pm 0.11$°C, $n = 756$) compared to those at Calcasieu Pass ($\bar{x} = 28.9 \pm 0.13$°C, $n = 483$) (T-test: $n = 1239$, $t_{1237} = -3.579$, $p < 0.001$).

Salinity across all sample sites varied between 4.5 and 36.0 ppt. Patterns in monthly mean salinity across all study areas were similar to those for temperature (i.e. peaks in August) (ANOVA: $n = 1235$, $F_{6,\,1228} = 73.060$, $p < 0.001$). However, Calcasieu Pass values were more variable and displayed no clear pattern (Fig. 20). A slightly significant increasing trend in annual mean salinity was seen at Sabine Pass, but not at Calcasieu (Table 4). A statistically significant differences in salinity between the two study areas was detected during across all years sampled (t-test: n = 1235, t_{1233} = -4.867,

Table 4. Summary of trendline statistics for annual mean values of abiotic factors at and across study areas in the NW Gulf during 1993-2002.

	n	slope	R^2	F	p
Sabine Pass					
Water temperature	10	-0.0715	0.02	0.160	0.697
Salinity *	10	0.7104	0.43	5.970	0.040*
Water visibility	10	-0.0063	0.06	0.530	0.489
Calcasieu Pass					
Water temperature	8	-0.0962	0.03	0.150	0.708
Salinity	8	-0.0079	< 0.01	0.007	0.935
Water visibility	8	-0.0112	0.16	0.319	0.319
All Sites Combined					
Water temperature	10	-0.0606	0.01	0.100	0.760
Salinity	10	0.2866	0.08	0.700	0.428
Water visibility	10	-0.0078	0.12	0.320	0.320

* significant result at alpha = 0.05

Table 5. ANOVA statistics for annual differences in abiotic factors at and across study areas in the NW Gulf during 1993-2002.

	n	df	F	p
Sabine Pass				
Water temperature	756	9, 748	6.577	< 0.001
Salinity	752	9, 742	16.361	< 0.001
Water visibility	762	9, 752	5.632	0.001
Calcasieu Pass				
Water temperature	483	7, 475	29.649	< 0.001
Salinity	483	7, 475	23.907	< 0.001
Water visibility	478	7, 470	2.059	0.047
All Sites Combined				
Water temperature	1239	9, 1238	20.289	< 0.001
Salinity	1235	9, 1234	22.502	< 0.001
Water visibility	1240	9, 1239	3.196	< 0.001

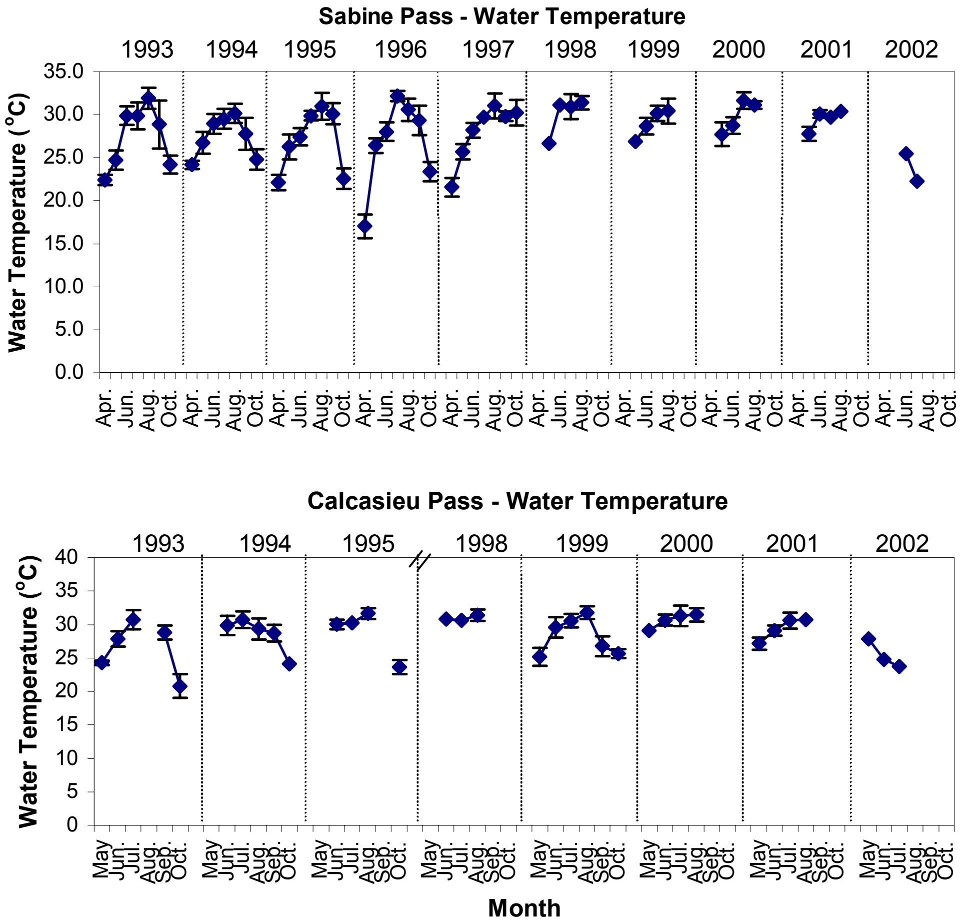


Figure 19. Time series of monthly mean water temperature (w/ standard devation bars) at Sabine and Calcasieu Passes during April-October 1993-2002 (Note: There are no data for Calcasieu in 1996 and 1997)

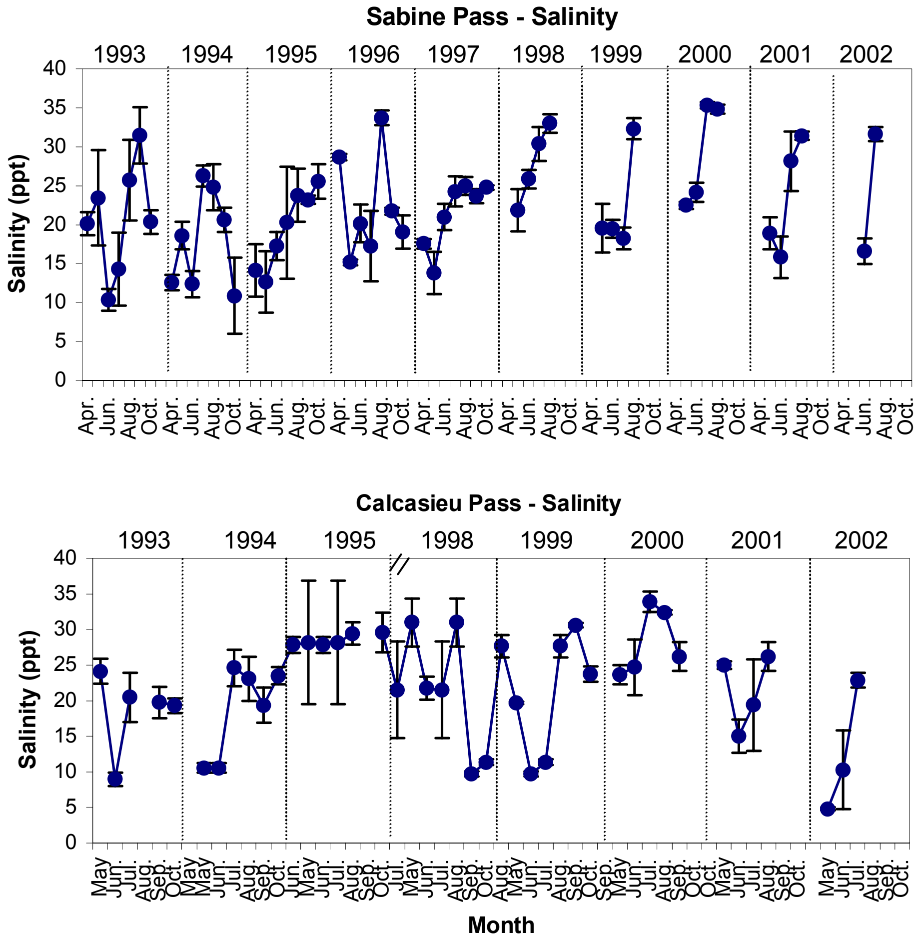


Figure 20. Time series of monthly mean salinity (w/ standard deviation bars) at Sabine and Calcasieu Passes during April-October 1993-2002 (Note: There are no data for Calcasieu in 1996 and 1997).

$p < 0.001$). Examination of rainfall data showed no significant trend over time (Upper TX coast: $n = 11$ slope = -0.6821, $F_9 = 0.070$, $r^2 < 0.01$, $p = 0.799$; SW LA coast: $n = 11$ slope = -0.2533, $F_9 = 0.008$, $r^2 < 0.01$, $p = 0.928$), but some variation between years. There was a significant difference in annual rainfall amounts between those for the upper Texas and southwestern Louisiana coasts (t-test: $n =11$, $t_{10} = -5.966$, $p < 0.001$), with the latter having a higher mean level ($\bar{x} = 159.3 \pm 8.1$ cm). However, both regions exhibited similar rainfall patterns (Pearson correlation: $r = 0.87$, $p = 0.001$), with precipitation lows in 1996 and 1999. Freshwater inflow to the Sabine-Neches estuary during 1993-1999 was highly variable between months, but with peaks in spring (April) and fall (September-October) (Appendix A). A significant low in annual freshwater inflow was observed in 1996 and attributed to widespread drought conditions in Texas (TWDB, 1999). All other years had inflow amounts well above the optimum ecological requirement of 11.8 million cubic km/year (TNRCC, 2002). Additionally, there was a very weak, but statistically significant, relationship between monthly mean salinity at Sabine Pass and monthly rainfall totals on the upper Texas (Linear regression: $n = 52$, $r^2 = 0.09$, $F_{1,51} = 4.707$, $p = 0.035$), but no relationship was detected between annual values ($n = 10$, $r^2 = 0.06$, $F_{1,8} = 0.481$, $p = 0.508$). Similarly, there was a stronger and statistically significant relationship between salinity and freshwater inflow at Sabine Pass on a monthly basis ($n = 52$, $r^2 = 0.32$, $F_{1,41} = 19.025$, $p < 0.001$), but not on an annual basis ($n = 10$, $r^2 = 0.19$, $F_{1,5} = 1.195$, $p = 0.324$). Peak annual mean salinity at Sabine Pass during 2000 coincided with severe drought conditions (NCDC, 2003).

All water visibility measurements at study sites in the NW Gulf were less than 2.0 m, and ranged from 0.08 to1.89 m. Mean visibility at Sabine and Calcasieu Passes exhibited no discernable trends on a monthly or annual basis (Fig. 21, Table 4) (p > 0.05), and no statistical difference was detected between study areas (t-test: n =1240,

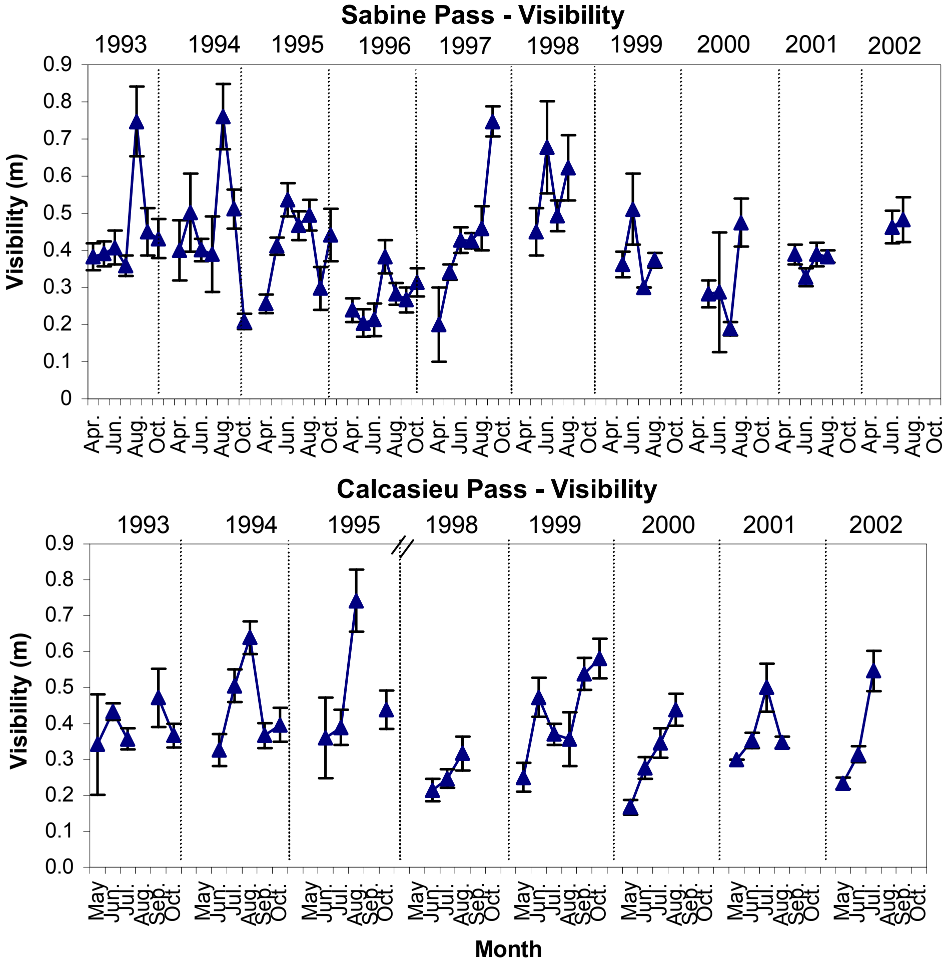


Figure 21. Time series of monthly mean visibility (w/ standard error bars) at Sabine and Calcasieu Passes during April-October 1993-2002 (Note: There are no data for Calcasieu in 1996 and 1997).

$t_{1238} = -1.243$, $p = 0.214$).

Analysis of the relationship between Kemp's ridley CPUE and each abiotic factor yielded no significant monthly or annual results (Table 6), with the exception of a negative relationship between log-transformed monthly ridley CPUE and monthly mean salinity, as well as, between annual mean ridley CPUE and salinity at Sabine Pass.

Table 6. Summary of least squares linear regression results for monthly and annual Kemp's ridley CPUE versus abiotic factors at and across study areas in the NW Gulf (1993-2002).

	Monthly Mean Abiotic & [Log(ridley CPUE+1)]				Annual Means			
	n	R^2	F	p	n	R^2	F	p
Sabine Pass								
Water temperature	52	< -0.01	0.423	0.518	10	-0.05	0.406	0.542
Salinity *	52	-0.18	10.638	0.002*	10	-0.64	14.275	0.005*
Water visibility	52	0.04	1.789	0.187	10	0.25	2.708	0.138
Calcasieu Pass								
Water temperature	34	0.02	0.754	0.392	8	-0.07	0.47	0.519
Salinity	34	< 0.01	0.017	0.897	8	< -0.01	0.002	0.967
Water visibility	34	0.03	0.959	0.335	8	0.08	0.493	0.509
All Sites Combined								
Water temperature	n/a	n/a	n/a	n/a	10	-0.07	0.584	0.467
Salinity	n/a	n/a	n/a	n/a	10	-0.25	2.612	0.145
Water visibility	n/a	n/a	n/a	n/a	10	0.36	4.455	0.068

* significant result at an alpha = 0.05

Discussion

The range of water temperatures recorded during the April-October sampling period is within documented physiological tolerance limits for the Kemp's ridley sea turtle (Moon *et al.*, 1997; Spotila *et al.*, 1997). Although water temperatures occasionally fell below 20°C (e.g. at Sabine Pass in April 1996), annual mean temperatures (limited to April-October measurements, depending upon year) were all ≥ 25°C. As such, ridley occurrence in nearshore waters during this study was probably not influenced greatly by monthly or annual mean water temperature, as evidenced by lack of a significant relationship between the two. A complete assessment of the effect of temperature on ridley abundance would require year-round sampling. The temporal scale of this study did not allow for temperature extremes that might induce ridley movements, although lowest overall catch rates were observed in October concurrent with the onset of the cool season.

The negative relationship between Kemp's ridley CPUE and monthly and annual mean salinity at Sabine Pass is interesting. Ridleys are a euryhalinic species capable of inhabiting regions of normal marine salinity and freshwater (at least for a short duration) (Ortiz *et al.*, 2000). Although annual mean salinity exhibited an increasing trend at Sabine Pass, individual measurements did not exceed 36.0 ppt and apparently were within the tolerance range for ridleys (Lutz, 1997). The data also suggest that rainfall on the upper TX coast and freshwater inflow to the Sabine-Neches estuary significantly explained nearshore salinity fluctuations on a monthly basis, but not annually. This is most likely due to the more immediate effect of monthly changes in rainfall and inflow on nearshore salinity that is not as evident on an annual scale. However, severe drought conditions during 2000 probably contributed to the peak in annual mean salinity (NCDC, 2003). Nevertheless, the decline in ridley CPUE at Sabine Pass may not be a direct response to rising salinity but to a change in some other factor(s), such as prey availability, or due to indirect effects of salinity on predators and prey. The effect of salinity, rainfall and freshwater inflow on ridley prey is addressed in the following chapter.

Water visibility did not have a strong influence on Kemp's ridley catch rates, probably because visbility was consistently low (< 2.0 m) and not notably different than the norm for this region (Britton and Morton, 1989). Although it was hypothesized that suspended particles would scatter light underwater and make it more difficult for ridleys to see the net, there was no evidence of higher CPUE corresponding with low visibility. It is likely that the Kemp's ridley's underwater visual acuity is similar to that of loggerheads or other shallow water benthic foragers (Bartol and Musick, 2001; Bartol *et al.*, 2002), and, as such, is specially adapted to turbid conditions. Conversely, the lack of a significant relationship between ridley capture rates and visibility may indicate that ridleys are less reliant on vision to sense prey, predators or obstacles in murky waters.

Even though there were few direct relationships between Kemp's ridley occurrence and selected abiotic factors at study areas in the NW Gulf, numerous indirect effects (e.g. the impact of drought on blue crab dynamics) may contribute to ridley aggregation or lack thereof. Other physical characteristics of the environment may exert greater influence on ridley abundance and distribution, but were not included or identified in this study. For example, Gulf circulation patterns are largely responsible for transporting ridleys from the nesting beach to developmental foraging habitat and may have a significant impact on overall distribution of ridleys prior to reaching coastal regions (Carr, 1980, 1987; Collard and Ogren, 1990). However, this parameter was not empirically examined in the present study. Future research may address this issue by analyzing geostrophic current vectors and velocities (in pelagic waters > 200 m depth) derived from satellite altimetry data and sea surface height anomalies (Polovina *et al.*, 1999; CCAR, 2003), thereby allowing researchers to map potential ridley dispersion routes during the “lost years”. Inclusion of more study locations, extended duration of sampling and tracking of ridleys may be necessary to gain a better understanding of Kemp’s ridley interaction with abiotic factors in nearshore waters.

CHAPTER IV

PREY AVAILABILITY

Introduction

Juvenile and subadult Kemp's ridley sea turtles utilize shallow coastal waters of the NW Gulf of Mexico as developmental feeding grounds (Ogren, 1989; Márquez, 1994). Because feeding impacts ridley growth, maintenance and reproductive condition, examination of factors affecting foraging success could provide information relevant to the conservation and management of this endangered species (Bjorndal, 1997). The abundance and distribution of Kemp's ridleys in nearshore habitat may be largely dependent upon prey availability, and, as such, this factor is included as a major component in the conceptual model of Kemp' ridley occurrence presented in Chapter I (Fig. 2, Table 1). The specific portion of the model that pertains to this influence is isolated in Fig. 22.

Crabs have been identified as the preferred prey of both juvenile and adult Kemp's ridleys, and the species of crab consumed is generally related to the prevalence of crab species in a region (Liner, 1954; Dobie *et al.*, 1961; Hildebrand, 1982; Ogren, 1989; Shaver, 1991; Burke *et al.*, 1994; Werner, 1994). Dietary analyses of ridleys from the Gulf of Mexico documented a preference for portunid (swimming) crabs in the genera *Callinectes* and *Ovalipes* (Liner, 1954; Hildebrand, 1982; Ogren, 1989; Shaver, 1991; Werner, 1996; Schmid, 1998), whereas fecal and intestinal analysis of juvenile ridleys from coastal waters of New York and New England included a prevalence

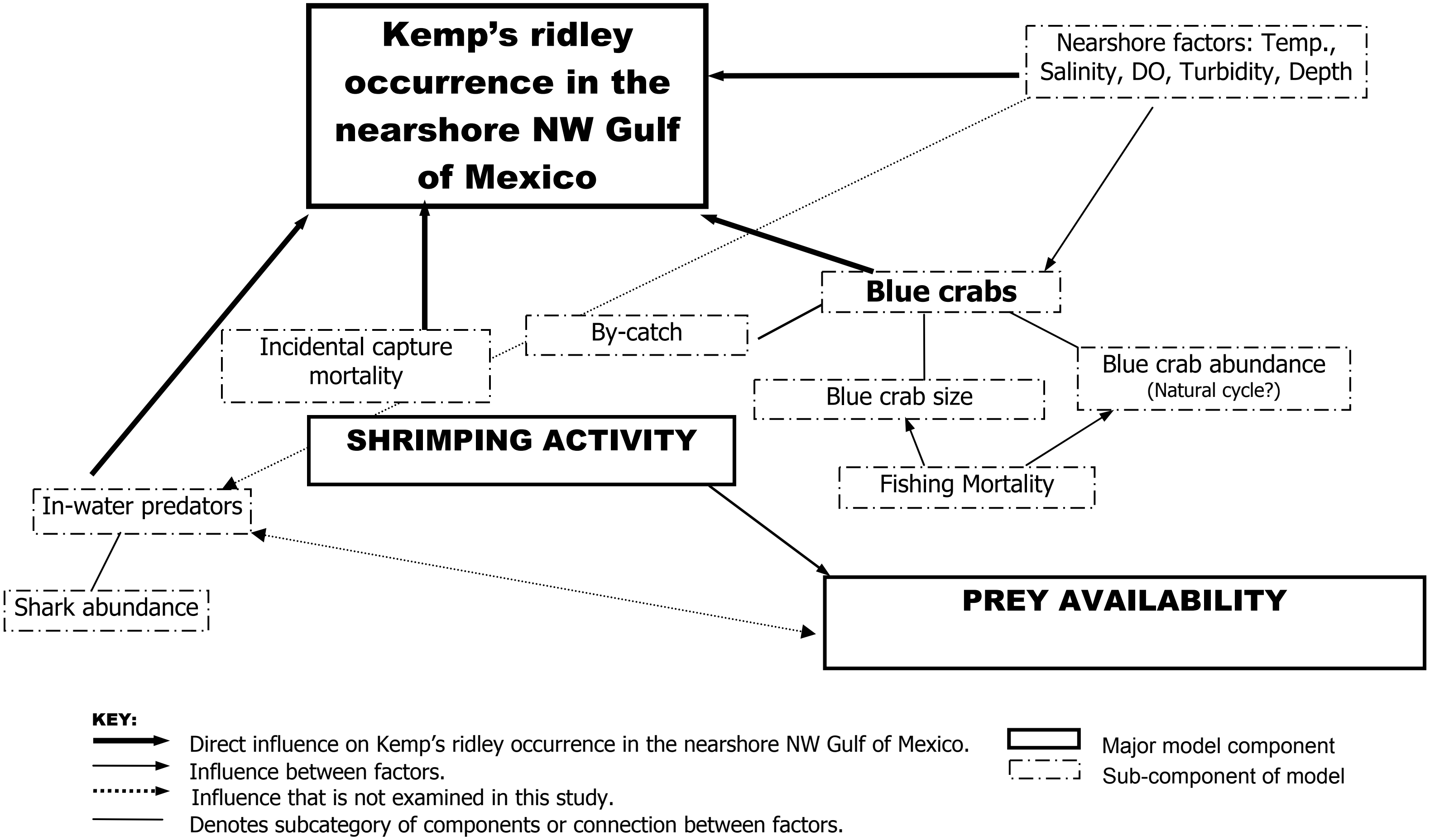


Figure 22. Portion of conceptual model detailing the hypothesized influence of prey availability on Kemp's ridley occurrence in nearshore waters of the NW Gulf of Mexico.

of walking crabs in the genera *Libinia* (spider crabs) and *Cancer* (rock crabs) in the diet (Dobie *et al.*, 1961; Burke *et al.*, 1994). Even within the Gulf of Mexico, there are slight differences in preferred crab species, most likely related to crab abundance and distribution. Shaver (1991) found that the speckled crab (*Arenaeus cribarius*) dominated the diet of wild sub-adult and adult Kemp's ridleys from south Texas waters, while Werner (1994) reported blue crabs (*Callinectes sapidus*) were most prevalent in the diet of wild juvenile and headstart ridleys captured in nearshore waters off the upper Texas and western Louisiana coasts. Schmid (1998) reported blue crabs and stone crabs (*Menippe spp.*) were dominant in the diet of Kemp's ridleys from the Cedar Keys, FL area. It is unknown how population density or size of blue crabs influences ridley abundance and distribution in the NW Gulf of Mexico. In order to examine this question, it is first necessary to address aspects of blue crab population dynamics and the fishery that exploits this species that may influence prey availability for ridleys.

Blue Crab Dynamics

Blue crabs are an estuarine-dependent species most commonly found in coastal waters of the Gulf of Mexico and southeast Atlantic (north of 28° N latitude). Their natural range extends from Nova Scotia to northern Argentina, including Bermuda and the Antilles (Perry and McIlwain, 1986). Blue crabs inhabit a variety of nearshore and estuarine habitats, with each serving the physiological requirements of one or more history stages (Perry and McIlwain, 1986). Spawning females are usually found in the lower estuary and adjacent nearshore waters because crab larvae require salinities in excess of 20 ppt for proper development. Conversely, adult male crabs prefer less saline water and thus are more common in upper reaches of the estuary. Saltmarsh fringe and vegetated bottom throughout the estuary have been identified as essential nursery habitat for juvenile crabs of both sexes (Thomas *et al.*, 1990)

Mating in northwestern Gulf estuaries typically occurs in low salinity waters during April-June and September-October. The seasonal occurrence of ridleys in nearshore waters during summer months coincides with this period of blue crab spawning, suggesting that foraging ridleys may be attracted to presence of larger gravid females.

Juvenile and adult blue crabs tolerate a wide range of salinities and water temperatures. Even though adult males, spawning females, and juveniles display preferences for differing salinities, they are euryhalinic and often inhabit salinities ranging from 0 to 40 ppt (Copeland and Bechtel, 1974). They also are found in water temperatures between 0 and 40°C, with increased growth rate of juveniles and female size at maturity linked to higher water temperature (Perry and McIlwain, 1986; Fisher, 1999). Freshwater inflow also has been identified as contributing to greater crab abundance (More, 1969; Guillory, 2000). However, this is more likely related to indirect effects on blue crab stocks, with greater overall estuarine productivity and food availability linked to higher freshwater inflow, as well as reduced presence of various predators in lower salinity waters (Wilber, 1994; Guillory, 2000).

There has been recent concern over the influence of bottom hypoxia (dissolved oxygen content < 2 mg/L) on movement and distribution of benthic invertebrates in the

Gulf of Mexico. The largest zone of oxygen-depleted waters along the US coast is located in shallow depths (5-60 m) over the Louisiana/Texas continental shelf and has been related to elevated nutrient discharge from the Mississippi River (Rabalais and Turner, 2001). Craig *et al.* (2001) proposed that sea turtle foraging in the NW Gulf might be impacted by hypoxia affecting abundance and distribution of benthic prey. Field studies and laboratory experiments have shown that blue crabs are sensitive to hypoxic conditions and will move to avoid DO levels less than 2 mg/L (Pihl *et al.*, 1991; Das and Stickle, 1993, 1994; Rabalais *et al.*, 2001). This response may result in a concentration of crabs, as well as other demersal organisms, shoreward of the hypoxic zone. As such, foraging Kemp's ridleys may encounter a greater density of prey in shallow Gulf waters during this period. Alternatively, benthic organisms may disperse east or west of the hypoxic zone resulting in lower crab abundances in nearshore waters along the Louisiana coast.

Blue crabs serve as prey for a number of vertebrate predators (Guillory and Elliot, 2001). The Kemp's ridley is one of several reptilian predators documented to feed on blue crabs (including the loggerhead sea turtle, diamondback terrapin - *Malaclemys terrapin*, and American alligator - *Alligator mississippiensis*), while red drum (*Sciaenops ocellatus*) has been ranked as the dominant ichthyofaunal consumer based on an index comparing all estuarine fish predators (Guillory and Elliot, 2001).

Human Impacts to Blue Crab Stocks

The commercial and recreational harvest of blue crabs in the Gulf of Mexico is a considerable fishery, averaging 29% of US blue crab landings during the 1990s (annually averaging of 28.2 million kg valued at $29.6 million) (Guillory *et al*., 1998). Louisiana consistently produces significantly higher annual blue crab landings (Guillory and Perret, 1998) than does any other Gulf state (72.7 % of total Gulf production, averaging 20.1 million kg in landings valued at $22.4 million annually during the 1990s), and even led the nation in 1987, 1988, 1991, 1992 and 2000 (Guillory, 2002). Florida and Texas rank second and third to Louisiana, respectively, in annual Gulf blue crab landings. Blue crab harvest in Texas is the state's third most valuable commercial fishery behind shrimp and oysters (Hammerschmidt *et al.*, 1998), with highest crab productivity in bays receiving the most freshwater (upper TX coast) (More, 1969). However, despite the seemingly high productivity, blue crab landings in Texas have declined by about 40% since 1987 (Wilson, 2001). A declining trend also has been reported in commercial CPUE and fishery-independent data for Texas and Louisiana (Guillory, 1997; Hammerschmidt *et al.*, 1998; Guillory and Perret, 1998; Guillory, 2002). In addition, there has been a reduction in the abundance of adult, legal sized crabs (> 127 mm carapace width), resulting in a decline in the average body size of the population. This declining trend in abundance of legal sized crabs has been linked to excessive fishing pressure on larger individuals or "growth overfishing" (Hammerschimdt *et al.*, 1998).

Other human-related impacts to the blue crab population include habitat degradation, ghost fishing by derelict traps and incidental capture in the shrimp fishery. Increased coastal development and altered flow regimes have resulted in loss of saltmarsh habitat and submerged aquatic vegetation that are essential to the growth and survival of juvenile blue crabs. Coastal pollution also may contribute to chemical contamination in blue crabs (Engel and Thayer, 1998). Contamination of the Kemp's ridley's primary prey may affect population recovery due to potentially harmful effects of bioaccumulation of toxins on this species' health and future reproductive success. Ghost fishing, the capture of crabs in lost or abandoned traps, has become a concern in recent years due to potentially important losses to the crab population. Guillory (1993) found that a substantial number of blue crabs die (25/trap) in abandoned or lost crab traps each year, with larger crabs less likely to escape these traps. The incorporation of biodegradable panels in crab traps and ghost trap retrieval programs have begun to address this problem.

Shrimping By-catch

Shrimp fishery by-catch poses another threat to blue crab stocks and a factor that may influence attractiveness of nearshore Gulf waters as Kemp's ridley foraging habitat. Ridley developmental feeding grounds in the NW Gulf overlap with areas consistently having the highest shrimping effort expended in US waters. It is speculated that sea turtles are attracted to by-catch discarded by shrimpers, and are at risk to incidental capture in trawls. By-catch is mostly composed of dead or dying animals, and is presumably an attractive food source for Kemp's ridleys. Presence of shrimp and fish in the stomach contents of ridleys lends evidence to Kemp's ridley foraging on by-catch, because ridleys are considered too slow to prey on these organisms (Magnuson *et al*., 1990; Shaver, 1991; Werner, 1994). Blue crabs also are an important component of by-catch. Sparks' (2000) assessment of by-catch associated with the inshore Matagorda Bay, Texas commercial shrimp fishery reported that blue crab comprised 10.5 % by weight of organisms incidentally captured in the spring shrimping season. Blue crab ranked fifth in by-catch biomass and abundance among 22 dominant species ($\geq$ 5% of a monthly total).

Relationship of *L. kempii* to Prey Availability

This chapter assesses the influence of prey availability on Kemp's ridley abundance and distribution in the NW Gulf. Such an assessment is necessary for understanding Kemp's ridley's use of nearshore developmental foraging habitat and how changes in foraging opportunities impact this species' growth, age at sexual maturity, fecundity and population recovery. Given the ridley's diet of portunid crabs, its attraction to by-catch and co-occurrence in areas of peak blue crab and shrimping density, this assessment will first characterize blue crab stocks and their limiting factors, as well as shrimping activity within the study area. I hypothesize that higher ridley CPUE will coincide with greater blue crab size and abundance and with increased availability of by-catch resulting from elevated shrimping activity.

Materials and Methods

Study areas and Kemp's ridley in-water survey methods are described in Chapter II. Measures of blue crab abundance and size were obtained via the following sources: 1) STFERL data collected in conjunction with netting operations; and 2) fishery-independent data obtained from the Texas Parks and Wildlife Department (TPWD) and Louisiana Department of Wildlife and Fisheries (LDWF).

Blue Crab Stocks: STFERL Data

Monthly blue crab abundance at each netting site was determined via replicate otter trawl tows. The otter trawl had a 6.1-m wide mouth, 19-mm bar mesh throughout, and 6.3-mm bar mesh in the cod end liner. Three replicate, 5-minute trawl tows were conducted parallel to the beachfront at each netting site during 1993-2002. During the 2001 sampling period, three additional trawl tows were conducted in deeper water approximately 1000 m offshore from the netting sites. These additional tows provided data for a comparison of blue crab abundance in shallow versus deeper water habitats. Each trawl sample was fixed in 10% formalin and transported to the lab for later analysis. Blue crabs were sorted from each trawl sample, enumerated, and weighed (g). Carapace width (mm) also was measured to determine size composition. Blue crab CPUE was calculated as the number of crabs per 5-minute trawl tow. Some months and stations were not sampled due to unfavorable weather conditions and varying research priorities. Inconsistency in the trawl crab data was considered when analyzing and interpreting results.

Crabs captured incidently in entanglement nets also were enumerated and included in the analysis, with resultant CPUE from this gear type calculated in the same manner as that for ridleys. Entangled crabs provided an additional measure of blue crab abundance, especially for larger individuals that may attract Kemp's ridleys foraging near netting locations. Carapace width of crabs captured in entanglement nets was not included in analyses due to size bias associated with gear selectivity (only large crabs were captured).

In addition, blue crab abundance from trawl and entanglement samples was assessed in relation to abiotic factors measured at sampling locations. Procedures for measuring water temperature (°C), salinity (ppt) and visibility (m) are described in Chapter III. Dissolved oxygen content (mg/L) also was included in these assessments and was recorded in the same manner as water temperature and salinity (with a YSI meter, thrice daily). Additional information on dissolved oxygen content, in the form of estimated areal extent (km^2) of the hypoxic zone since 1985, was obtained from Dr. Nancy N. Rabalais at the Louisiana Universities Marine Consortium (LUMCON). Rainfall (cm) and freshwater inflow (cubic km/year) data were gathered from agencies, as reported in Chapter III, to assess their possible influence on blue crab productivity (More, 1969, Guillory, 2000).

Blue Crab Stocks: TPWD and LDWF Fishery-Independent Data

Blue crab fishery-independent monitoring surveys conducted by the TPWD and LDWF were expressed as CPUE. Only trawl surveys from Gulf monitoring stations near ridley sample sites were used in analyses, so that comparisons could be made with STFERL data. Dr. Mark Fisher of the TPWD provided annual CPUE and mean carapace width (mm) for blue crabs collected in Gulf trawls near Sabine Pass during 1986-2001. These trawls were conducted in Texas territorial waters (16.7 km offshore and within 24.1 km of Sabine Pass) using a 6.1 m (at mouth) otter trawl with 3.8 cm stretch mesh throughout. Annual (including all months) trawl crab CPUE was calculated from 16 samples/month and expressed as number of crabs captured per hour.

Vince Guillory of the LDWF Marine Fisheries Division provided fishery-independent trawl data for Gulf stations near Calcasieu Pass during 1992-2002 (station 90 = 29° 45.0'N, 93° 20.0'W and station 91 = 29° 44.0'N, 93° 22.0'W). Crab samples were collected via a 4.9 m (16-foot) flat otter trawl (without tickler chain and with 19.0 mm and 6.4 mm bar mesh in body and cod end, respectively) towed for 10 minutes (Guillory, 1997). Blue crab CPUE was expressed as number of crabs/10-minute tow, while carapace width of up to 50 individuals in each trawl was measured in 5-mm intervals.

Shrimping Activity/By-catch

Effect of shrimping activity/by-catch on Kemp's ridley abundance in the study area was assessed in three ways. One method compared Kemp's ridley CPUE from west-side stations at Sabine Pass (TX waters) before, during and after the Texas Closure to shrimping (mid-May to mid-July). The second method compared ridley CPUE from west-side stations at Sabine and Calcasieu Passes during the Texas Closure to ascertain differences in catch rate between areas with (Calcasieu Pass) and without (Sabine Pass) shrimping activity. A third method for assessing influence of shrimping activity/bycatch on ridley occurrence was conducted over all locations during the 2001-2002 sampling years only. During these years, number of shrimp boats within sight of netting operations was recorded to compare: 1) days with ridley capture and shrimping activity; 2) days without ridley capture but with shrimping activity; 3) days with ridley capture but no shrimping activity; and 4) days with no ridley captures and no shrimping activity.

Data Analysis

Monthly blue crab CPUE from trawl samples was calculated by summing the number of crabs collected and dividing by total number of trawl tows. Monthly entanglement crab CPUE was calculated in the same manner as ridley CPUE (expressed as number/km-hour). Monthly CPUE values were then log transformed [Log (CPUE+1)] to approximate a normal distribution. One-way ANOVA was used to identify differences in blue crab CPUE and carapace width (mm) between years (across all sites combined and at Sabine and Calcasieu Passes), while a t-test was used to examine differences in these parameters between study areas (Sabine and Calcasieu Passes) and blocks of years (1993-1997 vs. 1998-2002). One-way ANOVA also was used to detect differences in blue crab CPUE between sample months (April-October) and netting stations (1, 3, 4, 5) for all study sites combined. Post hoc multiple comparisons were conducted using the Least Significant Difference (LSD) test. Sampling biases described in Chapter II were also taken into consideration when interpreting differences in monthly shark CPUE between netting stations. Trends in annual mean crab abundance and size were assessed via the same method described in previous chapters (null hypothesis: slope of the trendline = 0). Pearson Correlation was utilized in comparing the similarity between fishery-independent trawl data from STFERL, TPWD, and LDWF (variables were not assessed as independent and dependent). Least squares linear regression analysis was used to examine relationships between Kemp's ridley CPUE and blue crab abundance/size on both a monthly and annual basis. A best-fit line was then fitted to the data to produce the most appropriate representation of the relationship between independent and dependent variables. Annual mean blue crab CPUE from entanglement nets was transformed (-1/x) to produce a linear relationship when plotted against annual mean Kemp's ridley CPUE (Ott, 1993). Multiple regression was used to assess the influence of both annual mean blue crab CPUE and size from STFERL trawl samples on Kemp's ridley CPUE at Sabine Pass and across all sites combined.

One-way ANOVA was used to identify differences in DO (mg/L) between years (across all sites combined and at Sabine and Calcasieu Passes), while a t-test was used to examine differences in this parameter between study areas (Sabine and Calcasieu Passes). ANOVA also was used to detect differences in DO between sample months (April-October). Least squares linear regression was used to examine the relationship between blue crab CPUE and abiotic factors (water temperature, salinity, visibility, and DO) on a monthly and annual basis. The relationship between blue crab CPUE and freshwater inflow at Sabine Pass also was assessed via least squares linear regression. Because there may a time lag on the order of several months to a year between river discharge and its impact on blue crab abundance (Guillory, 2000), the relationship between blue crab CPUE and freshwater inflow also was analyzed with a 1-yr lag.

One-way ANOVA with LSD post hoc multiple comparisons was used to assess Kemp's ridley CPUE before, during and after the Texas Closure at Sabine Pass. Difference in ridley abundance at Sabine Pass (TX waters) vs. Calcasieu Pass (LA waters) during the closure was examined via a paired t-test of annual CPUE for comparable years. A Chi-square Test of Independence was used to assess the association between ridley capture and shrimping activity based on: 1) sampling days with ridley capture and shrimping boats present; 2) days with no captures but shrimping boats present; 3) days with ridley capture but no shrimping activity; and 4) days with no captures and no shrimp boats present.

Mean values in this chapter are expressed as $\bar{x}$ ± standard error. All statistical analyses were conducted using an α level of 0.05 in Microsoft Excel and SPSS statistical software packages.

Results

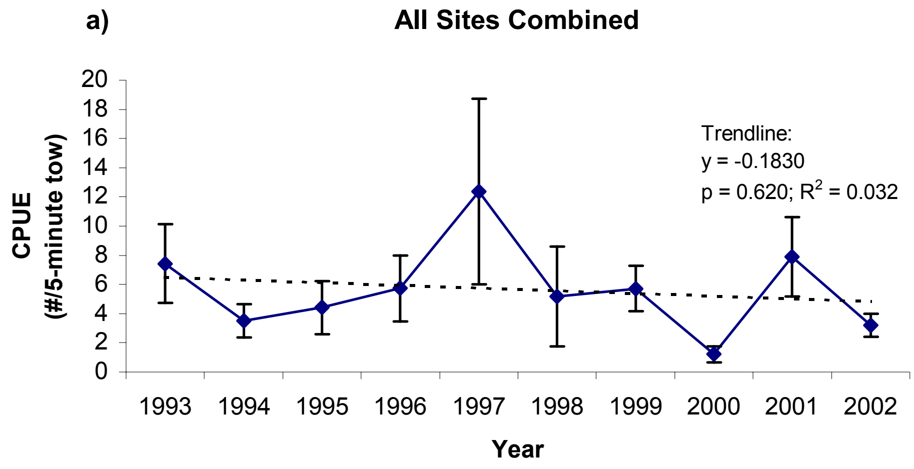
Blue Crab Stock Assessment

STFERL Data: Trawl Samples

Annual mean blue crab CPUE from STFERL trawl samples at all sites combined in the NW Gulf fluctuated among years, with no significant trend ($n = 10$, slope = -0.1853, $r^2 = 0.03$, $F_8 = 0.270$, $p = 0.620$) (Fig. 23a). Mean blue crab abundance peaked in 1997 (12.4 ± 6.4 crabs/5-min tow, $n = 5$), and fell to a study-wide low in 2000 (1.2 ±

0.55 crabs/5-min tow, $n = 6$). However, there was only a slight statistically significant difference between years (ANOVA: $n = 70$, $F_{9,60} = 2.020$, $p = 0.055$). LSD post hoc multiple comparisons revealed that CPUE in 2000 was significantly lower than that in 1993 ($p = 0.013$), 1997 ($p = 0.028$), 1999 ($p = 0.002$) and 2001 ($p = 0.004$). Also CPUE in 1997 was significantly greater than 1994 ($p = 0.035$) and 1995 ($p = 0.019$).

Blue crab CPUE from trawls at respective study areas is displayed in Figure 23b. CPUE at Sabine Pass exhibited no statistically significant trend ($n = 10$, slope = -0.6473, $r^2 = 0.27$, $F_8 = 3.01$, $p = 0.121$) and was not significantly different between years (ANOVA: $n = 45$, $F_{9,35} = 1.430$, $p = 0.213$). The Calcasieu Pass counterpart also yielded no significant trend ($n = 8$, slope = 0.6176, $r^2 = 0.19$, $F_6 = 1.360$, $p = 0.288$), but a peak was observed in 2001 (16.3 ± 2.4 crabs/ 5-min tow, $n = 4$). This elevated CPUE resulted in a significant difference between years (ANOVA: $n = 25$, $F_{7,17} = 2.690$, $p = 0.045$) LSD post hoc comparisons revealed CPUE in 2001 was significantly greater than that in 1993 ($p = 0.013$), 1994 ($p = 0.029$), 1995 ($p = 0.002$), and 2000 ($p = 0.012$). No significant difference in blue crab CPUE was detected between Sabine and Calcasieu Passes over the entire study period (T-test: $n = 70$, $t_{68} = 1.360$, $p = 0.178$).



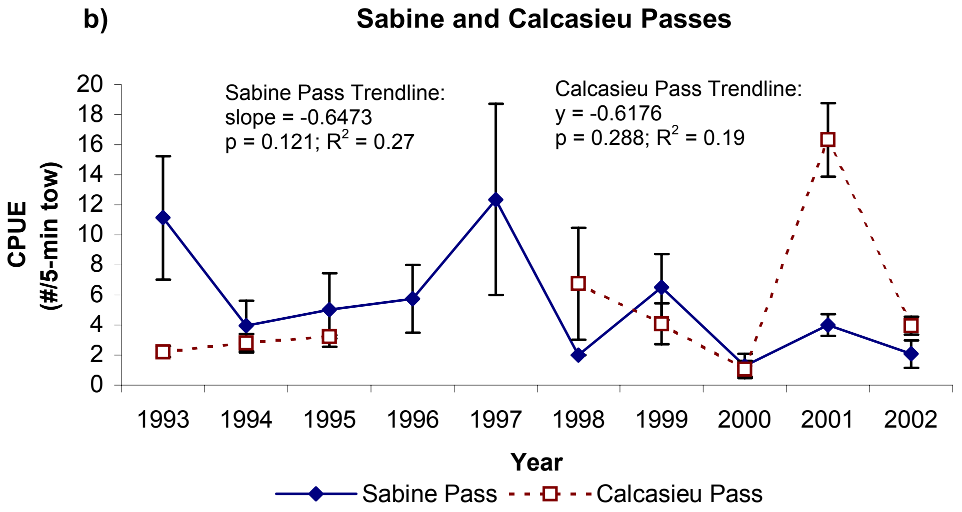


Figure 23. Annual mean blue crab CPUE (#/5-minute tow) from trawl samples collected across all sites combined (a) and at Sabine and Calcasieu Passes (b) during 1993-2002.

However, a difference was detected during 1993-1995 (n = 32, $t_{30} = 2.372$, p = 0.024), with Calcasieu Pass having higher crab CPUE. There was no statistical difference between the two study areas during 1998-2002 (n = 28, $t_{26} = -1.249$, p = 0.223) despite a large peak in CPUE at Calcasieu Pass in 2001.

A comparison of blue crab CPUE from trawl samples taken west and east of the jetties yielded no significant difference across NW study areas ($n = 127$, $t_{125} = 1.264$, $p = 0.209$). There also were no CPUE differences between shallow (< 3 m) and deeper water stations (3-5 m) during the 2001 sampling season ($n = 16$, $t_{14} = -0.961$, $p = 0.353$).

Mean size of blue crabs declined significantly across all locations in the NW Gulf ($n = 10$, slope = -6.339, $r^2 = 0.72$, $F_8 = 20.10$, $p = 0.002$) and at respective study areas (Sabine Pass: $n = 10$, slope = -5.410, $r^2 = 0.50$, $F_8 = 7.99$, $p = 0.022$; Calcasieu Pass: $n = 8$, slope = -6.777, $r^2 = 0.70$, $F_6 = 14.15$, $p = 0.009$) (Fig. 24). Comparison of mean blue crab carapace width (mm) at Sabine and Calcaseiu Passes yielded significantly larger crabs at Sabine Pass when analyzing across all years (t-test: $n = 2795$, $t_{2693} = 11.035$, $p < 0.001$). However, a comparison of blue crab size at the study areas during blocks of years yielded significantly larger crabs at Calcasieu Pass during 1993-1995 ($n = 1211$, $t_{1209} = -4.198$, $p < 0.001$), and an opposite result during 1998-2002, when larger conspecifics were present at Sabine Pass ($n = 956$, $t_{954} = 3.245$, $p = 0.001$)

STFERL Data: Entanglement Net Samples

Blue crab abundance in entanglement nets across all NW Gulf sites and at respective study areas displayed a great deal of monthly and annual variability (Table 7), with significantly larger annual mean CPUE observed in 1994 (18.86 ± 5.40/ km-hr, n = 12) and 1996 (30.50 ± 14.19/ km-hr, n = 7) (ANOVA: $n = 86$, $F_{9,76} = 3.855$, $p = 0.001$) (Fig. 25). Other years exhibited much lower mean values ranging from 1.70 to 9.28

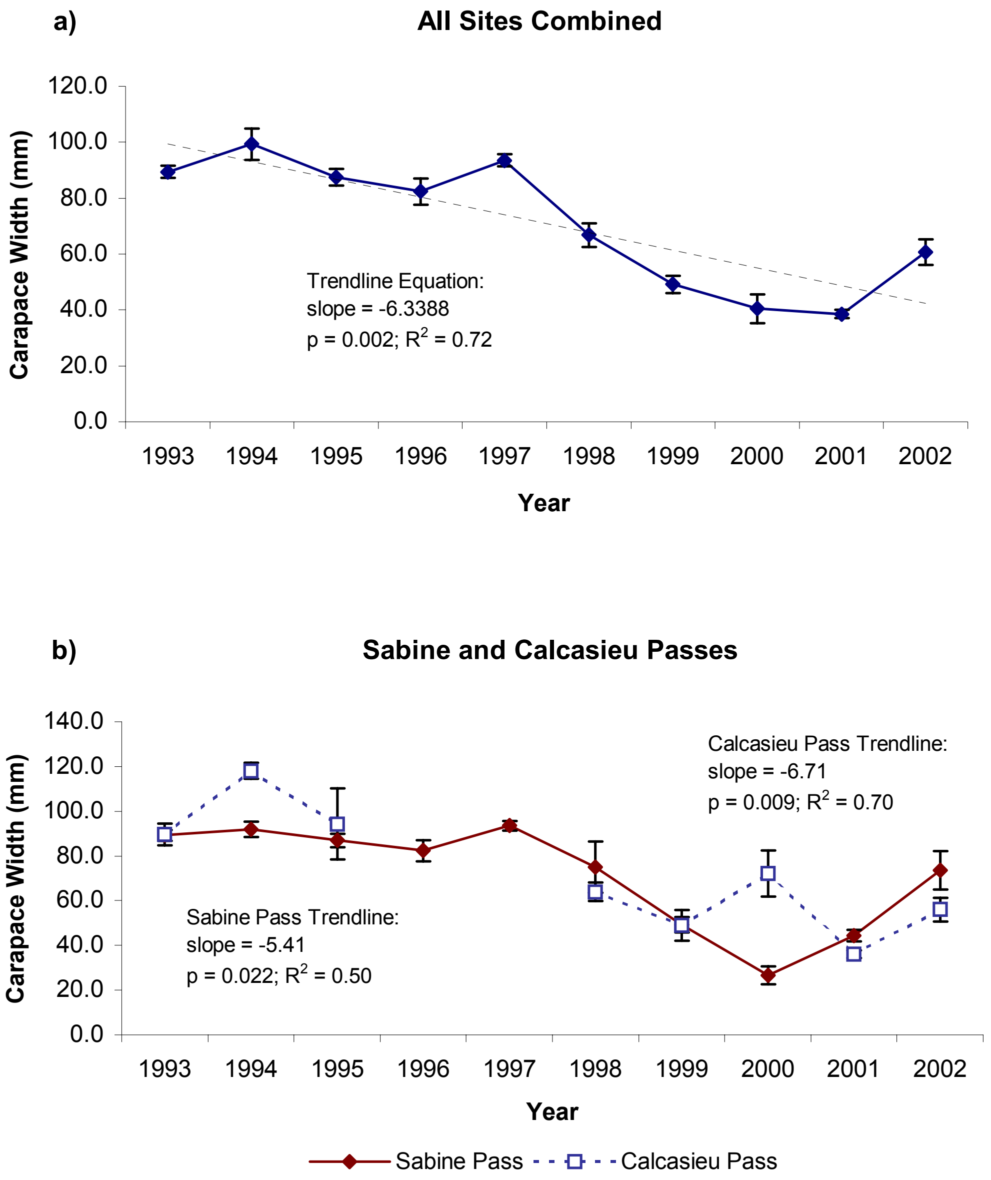


Figure 24. Annual mean blue crab carapace width (mm) (w/ standard error bars) from trawl samples collected across all sites combined (a) and at Sabine and Calcasieu Passes (b) during 1993-2002.

Table 7. Monthly blue crab CPUE from entanglement nets at Sabine and Calcasieu Passes, and annual CPUE statistics for all sites combined in the NW Gulf during 1993-2002.

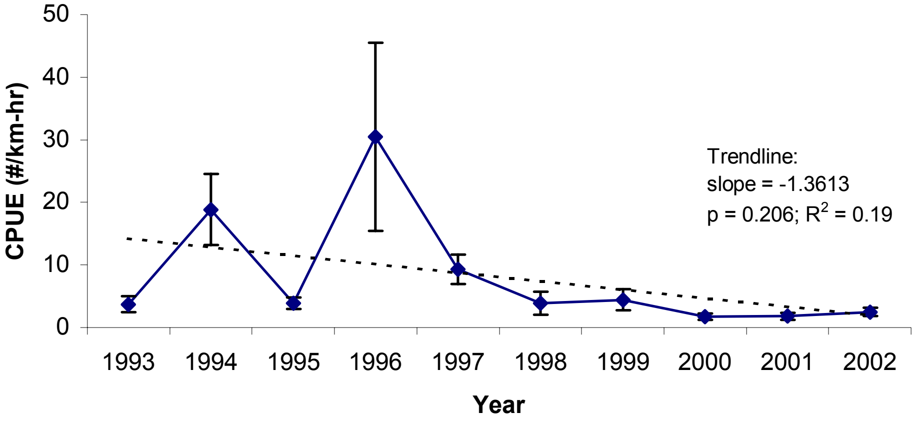
	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Sabine Pass										
April	0.50	22.83	7.06	8.58	14.38					
May	4.33	40.77	4.78	24.73	12.88	0.79	1.23	1.96	1.65	
June	4.98	40.78	7.68	92.66	2.36	0.99	15.38	1.35	3.19	0.47
July	12.92	14.69	6.67	75.06	15.67	2.82	8.68	0.19	1.85	2.42
August	0.91	1.24	0.18	2.96	4.25	0.56	0.36	0.00	0.16	
September	0.31	3.81	1.41	7.19	6.14					
October	1.56	6.02	0.00	2.34						
n	476	735	177	2514	714	82	254	22	63	26
Mean	**3.65**	**18.59**	**3.97**	**30.50**	**9.28**	**1.29**	**6.41**	**0.88**	**1.71**	**1.44**
STD	4.49	16.81	3.36	37.55	5.70	1.03	7.05	0.94	1.24	1.38
STERR	1.70	6.35	1.27	14.19	2.16	0.39	2.66	0.35	0.47	0.52
CV	123.19	90.41	84.80	123.10	61.46	80.23	109.94	0.00	72.36	95.38
Calcasieu Pass										
May	1.59						0.78	3.38	0.80	3.49
June	5.19	48.67	6.67			10.95	5.35	4.08	4.91	1.75
July	11.22	40.18	4.87			10.87	9.31	2.00	1.56	4.36
August		3.37	0.62			0.14	2.54	0.64	0.47	
September	0.27	3.13					0.07			
October	0.81	0.79	2.66				0.76			
n	96	690	77			189	140	110	148	145
Mean	**3.81**	**19.23**	**3.71**			**7.32**	**3.14**	**2.52**	**1.94**	**3.20**
STD	4.56	23.22	2.63			6.22	3.58	1.52	2.04	1.33
STERR	2.04	10.38	1.18			2.78	1.60	0.68	0.91	0.60
CV	119.59	120.77	70.90			84.91	113.98	60.40	105.17	41.62
All Sites Combined										
n	572	1425	254	2703	854	192	402	132	211	171
Mean	**3.72**	**18.86**	**3.87**	**30.50**	**9.28**	**3.87**	**4.45**	**1.70**	**1.83**	**2.50**
STD	4.31	18.72	2.98	37.55	5.70	4.88	5.15	1.47	1.57	1.51
STERR	1.24	5.40	0.90	14.19	2.33	1.84	1.72	0.52	0.55	0.68
CV	115.99	99.25	76.96	123.10	61.46	125.96	115.82	86.22	85.76	60.54

Blank = Not Sampled

CPUE shown for Calcasieu 1999 and 2000 also includes that from Mermentau Pass, LA.

n = total number of individuals captured; STD = Standard Deviation; STERR = Standard Error; CV = Coefficient of Variation

a) All Sites Combined



b) Sabine and Calcasieu Passes

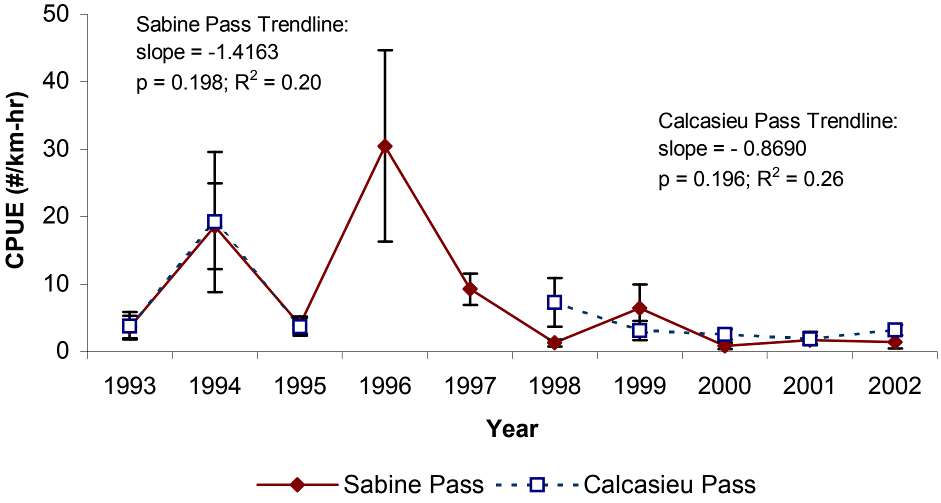


Figure 25. Annual mean blue crab CPUE from entanglement nets across all sites combined (a) and at Sabine and Calcasieu Passes (b) during 1993-2002.

crabs/km-hr. No statistically significant trend in blue crab CPUE was observed across NW Gulf sites ($n = 10$, slope $= -1.3613$, $r^2 = 0.19$, $F_8 = 1.89$, $p = 0.206$) or at Sabine and Calcasieu Passes (Sabine Pass: $n = 10$, slope $= -1.4163$, $r^2 = 0.20$, $F_8 = 1.98$, $p = 0.198$; Calcasieu Pass: $n = 8$, slope $=-0.8690$, $r^2 = 0.26$, $F_6 = 2.12$, $p = 0.196$). However, a difference in crab CPUE among years was detected at Sabine Pass (ANOVA: $n = 52$, $F_{9, 42} = 3.679$, $p = 0.002$), with CPUE in 1994 and 1996 significantly different from all other years except 1997, 1998, and each other ($p< 0.05$). Calcasieu Pass did not exhibit any statistically significant differences in CPUE between years (ANOVA: $n = 34$, $F_{7, 26} = 0.0859$, $p = 0.551$). Additionally, there was no difference in CPUE between Sabine and Calcasieu Passes across years (T-test: $n = 86$, $t_{84} = 0.962$, $p = 0.339$).

A comparison of monthly CPUE across sites and years yielded a significant difference between earlier (April-July) and later (August-October) sample months (ANOVA: $n = 86$, $F_{6, 79} = 5.291$, $p < 0.001$) (Fig. 26). Conversely, a comparison of entanglement crab CPUE at respective netting stations (across all locations in the NW Gulf) yielded no statistical differences in catch rate between west/east and jetty/beachfront locations (ANOVA: $n = 171$, $F_{3, 167} = 0.340$, $p = 0.749$).

Fishery-Independent Data: TPWD Trawl Samples

Annual blue crab CPUE (1986-2001) from TPWD trawl surveys conducted within 24.1 km of Sabine Pass is displayed in Figure 27. There was no significant trend present from 1986-2001 ($n = 17$, slope $= -0.1243$, $r^2 = 0.01$, $F_{14} = 0.180$, $p = 0.675$), but peaks were observed in 1991 (18.9/hr), 1994 (9.2/hr) and 1997 (15.6/hr) (M. Fisher, *personal communication*).

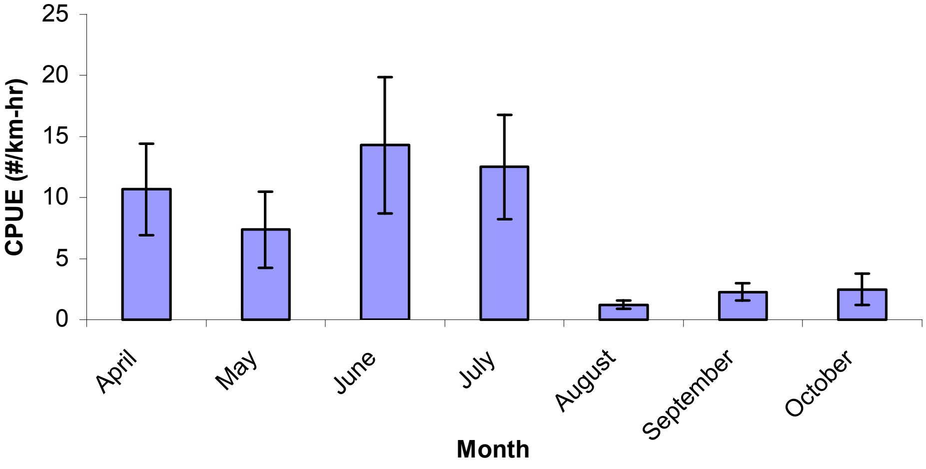


Figure 26. Mean blue crab CPUE for months entanglement netting occurred at study areas in the northwestern Gulf of Mexico during 1993-2002.

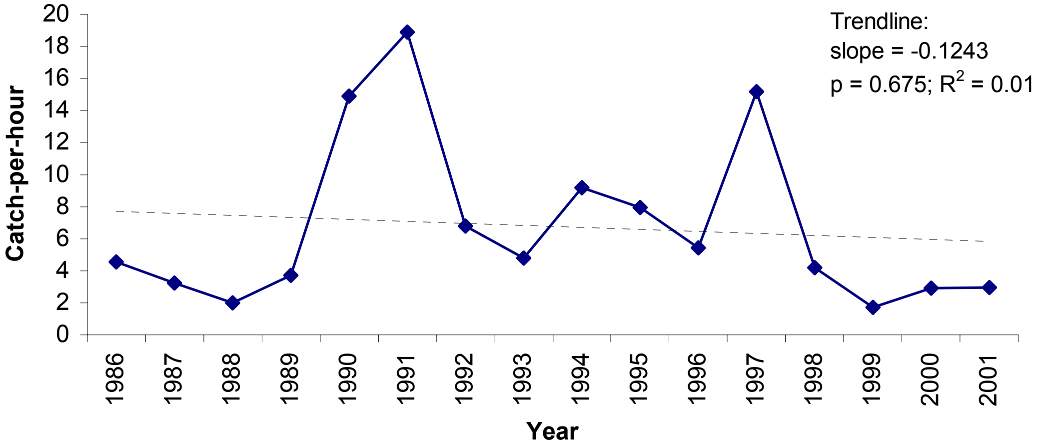


Figure 27. TPWD annual blue crab CPUE from trawl samples collected within TX state territorial waters (offshore to 16.7 km) and 24.1 km west of Sabine Pass (1986-2001; Data provided by M. Fisher).

There also was no strong correlation between these data and those from STFERL trawl tows (Pearson correlation: n = 9, r = 0.54, p = 0.125). Blue crabs in TPWD samples exhibited a significantly declining trend in size composition when including all years of data (n = 17, slope = -1.7573, $r^2 = 0.48$, $F_{14} = 12.69$, p = 0.003), but failed to yield a similar result for 1993-2001 (n = 9, slope = -1.360, $r^2 = 0.35$, $F_7 = 3.70$, p = 0.096) (Fig. 28).

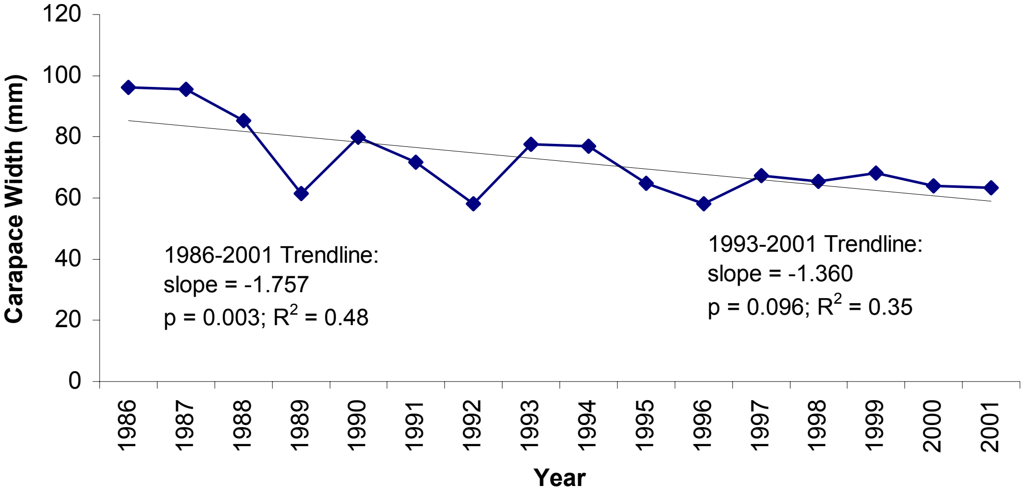


Figure 28. TPWD annual mean blue crab carapace width from trawl samples taken near Sabine Pass, TX (Data provided by M. Fisher; standard error was not included).

Fishery-Independent Data: LDWF Trawl Samples

Annual blue crab CPUE from LDWF fishery-independent trawl surveys (1992-2002) conducted near Calcasieu Pass declined, although the trend was not statistically significant (n = 11, slope = -1.0385, $r^2 = 0.23$, $F_9 = 2.66$, p = 0.138) (Fig. 29). Peaks in abundance occurred in 1993 (27.5/10-min. tow), 1997 (12.9/10-min. tow) and 2001 (10.1/10-min. tow), but there was no correlation with similar statistics from STFERL trawl samples collected at Calcasieu Pass (Pearson correlation: $n = 8$, $r = 0.10$, $p = 0.830$). There were also no discernable trends in blue crab size over the years, but the majority of crabs were < 80 mm carapace width.

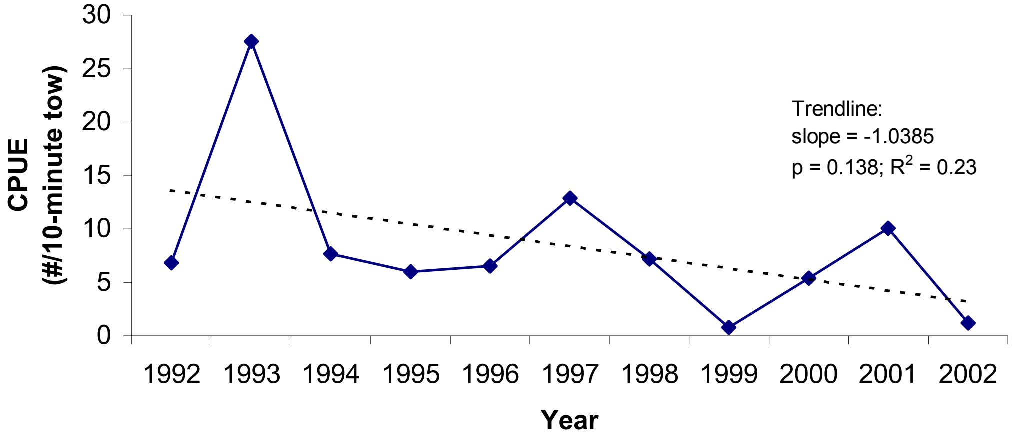


Figure 29. Annual blue crab CPUE from LDWF trawl samples collected in the Gulf waters near Calcasieu Pass, LA during 1992-2002 (Data provided by V. Guillory).

Environmental Influence on Blue Crab Stocks

Blue crab abundance was examined in relation to several abiotic factors. Analyses of water temperature, salinity, visibility, rainfall, and freshwater inflow for Sabine Pass only were reported in Chapter III. Annual mean dissolved oxygen content exhibited no particular trend over all sites combined ($n = 7$, slope = 0.0012, $r^2 < 0.01$, F_8

= 0.001, p = 0.990) but did significantly vary between years (ANOVA: n = 644, $F_{6, 643}$ = 8.203, p < 0.001). No significant trends in annual mean DO were observed at Sabine (n = 10, slope = -0.0652, r^2 = 0.04, F_8 = 0.210, p = 0.663) or Calcasieu Passes (n = 8, slope = -0.2965, r^2 = 0.70, F_6 = 7.15, p = 0.075), but both study areas exhibited significant differences in DO between years (ANOVA: Sabine Pass n = 366, $F_{6, 643}$ = 8.355, p < 0.001 and Calcasieu Pass n = 278, $F_{6, 643}$ = 6.213, p < 0.001). Monthly mean DO values were typically lowest during July-September, but the majority of DO values were still above 2 mg/L (Fig. 30), demonstrating that hypoxic conditions were not present in the study area. There also was no significant difference in DO between respective study areas (T-test: n = 644, t_{642} = 1.549, p = 0.111). Estimated size of the hypoxic zone in the Gulf of Mexico has been growing since 1985 (Appendix A), with a particular increase in 1993 following extensive floods in the midwestern US that resulted in elevated nutrient discharge from the Mississippi River (Rabalais *et al.*, 2002). Although the hypoxic zone was relatively small in 2000 (4,400 km^2) due to low river flow and nutrient flux during drought conditions (Rabalais *et al.*, 2002), bottom-hypoxia reached its greatest extent to-date in 2002 (22,000 km^2).

Regression analyses of blue crab CPUE and abiotic factors revealed only four statistically significant relationships (Tables 8 and 9). A negative relationship was found between annual mean trawl crab CPUE and DO, as well as between monthly trawl crab CPUE and mean salinity at Sabine Pass. There also was a negative relationship detected between annual mean entanglement crab CPUE and freshwater inflow. However, when the relationship between entanglement blue crab abundance and freshwater inflow was analyzed with a one-year lag to account for a possible delayed effect on crab abundance (Guillory, 2002), no significant relationship was found.

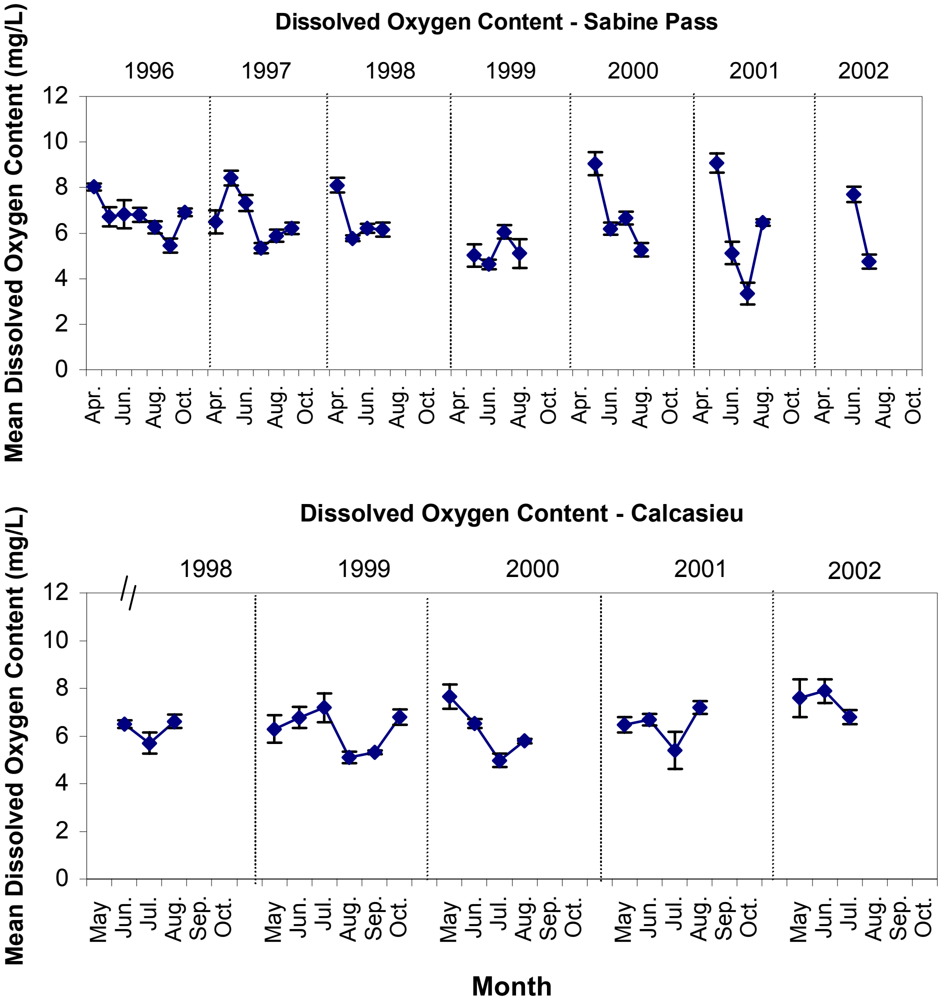


Figure 30. Time series of monthly mean dissolved oxygen (mg/L) (w/ standard error bars) at Sabine and Calcasieu Passes during April-October 1996-2002 (Note: There are no data for Calcasieu in 1996 and 1997).

Table 8. Summary of least squares linear regression results for monthly and annual blue crab CPUE from trawl samples versus abiotic factors at and across study areas in the NW Gulf (1993-2002).

	Monthly Mean Abiotic & [Log (crab CPUE+1)]				Annual Means			
	n	R^2	F	p	n	R^2	F	p
Sabine Pass								
Water temperature	45	-0.03	1.386	0.246	10	-0.03	0.222	0.650
Salinity *	45	-0.17	8.473	0.006*	10	-0.016	1.471	0.260
Water visibility	45	-0.03	1.225	0.275	10	-0.06	0.549	0.480
Dissolved oxygen	25	< 0.01	0.003	0.956	7	-0.64	8.828	0.031
Freshwater Inflow	n/a	n/a	n/a	n/a	9	-0.17	1.232	0.309
Inflow w/ 1 yr lag	n/a	n/a	n/a	n/a	9	0.26	2.053	0.202
Calcasieu Pass								
Water temperature	25	0.14	3.778	0.064	8	0.11	0.731	0.426
Salinity	25	-0.11	2.820	0.107	8	-0.06	0.353	0.574
Water visibility	25	-0.11	2.816	0.107	8	-0.04	0.273	0.620
Dissolved oxygen	13	-0.16	2.020	0.183	5	-0.11	0.359	0.591
All Sites Combined								
Water temperature	n/a	n/a	n/a	n/a	10	< 0.01	0.080	0.931
Salinity	n/a	n/a	n/a	n/a	10	-0.05	0.419	0.536
Water visibility	n/a	n/a	n/a	n/a	10	-0.04	0.219	0.604
Dissolved oxygen*	n/a	n/a	n/a	n/a	7	-0.06	7.383	0.042*

Note: Sabine Pass DO statistics are for 1996-2002;
Calcasieu Pass DO statistics are for 1998-2002
Sabine Pass Freshwater Inflow statistics are for 1993-1999.
* signifcant result at an alpha = 0.05

Table 9. Summary of least squares linear regression results for monthly and annual blue crab CPUE from entanglement nets versus abiotic factors at and across study areas in the NW Gulf (1993-2002).

	Monthly Mean Abiotic & [Log (crab CPUE+1)]				Annual Means			
	n	R^2	F	p	n	R^2	F	p
Sabine Pass								
Water temperature	52	-0.01	0.595	0.444	10	-0.04	0.302	0.598
Salinity *	52	-0.30	21.227	< 0.001*	10	-0.25	2.600	0.253
Water visibility	52	-0.07	3.606	0.063	10	-0.14	1.253	0.295
Dissolved oxygen	31	0.01	0.291	0.594	7	< 0.01	0.021	0.889
Freshwater Inflow *	42	< 0.01	0.159	0.692	7	-0.75	14.748	0.012*
Inflow w/ 1 yr lag					7	0.13	0.734	0.431
Calcasieu Pass								
Water temperature	34	0.17	3.245	0.091	8	0.02	0.097	0.766
Salinity	34	-0.18	3.526	0.079	8	< -0.01	0.006	0.939
Water visibility	34	-0.02	0.265	0.613	8	0.14	0.933	0.371
Dissolved oxygen	20	0.44	1.549	0.339	5	-0.05	0.169	0.708
All Sites Combined								
Water temperature	n/a	n/a	n/a	n/a	10	-0.04	0.361	0.564
Salinity	n/a	n/a	n/a	n/a	10	-0.10	0.871	0.378
Water visibility	n/a	n/a	n/a	n/a	10	-0.08	0.731	0.417
Dissolved oxygen	n/a	n/a	n/a	n/a	7	0.05	0.261	0.613

Note: Sabine Pass DO statistics are for 1996-2002;
Calcasieu Pass DO statistics are for 1998-2002
Sabine Pass Freshwater Inflow statistics are for 1993-1999.
* signifcant result at an alpha = 0.05

Kemp’s Ridley Relationship to Blue Crab Stocks

There were no strong significant relationships between Kemp’s ridley abundance and blue crab CPUE from trawl samples (Table 10), except versus TPWD annual data at Sabine Pass (Linear regression: $n = 9$, $r^2 = 0.63$, $F_{1,7} = 11.970$, $p = 0.011$) (Fig. 31). Regression analysis of ridley CPUE and crab size yielded a significant linear relationship over all sites ($n = 10$, $r^2 = 0.57$, $F_{1,8} = 10.848$, $p = 0.001$) and at Sabine Pass ($n = 10$, $r^2 = 0.76$, $F_{1,8} = 24.778$, $p < 0.001$), but not at Calcasieu Pass ($n = 8$, $r^2 = 0.23$, $F_{1,6} = 1.786$, $p = 0.230$) (Fig. 32). Multiple regression analysis of ridley CPUE with both size and abundance of trawled crabs indicated a significant relationship overall ($n = 10$, $r^2 = 0.76$, $F_{2,6} = 10.961$, $p = 0.007$), but blue crab size was a better predictor of Kemp’s ridley occurrence at Sabine Pass (Table 11). There also was a strong linear relationship between annual mean Kemp’s ridley abundance and transformed (-1/x) entangled crab CPUE across all sites in the NW Gulf ($n = 10$, $r^2 = 0.63$, $F_{1,8} = 13.819$, $p = 0.006$) (Fig. 33), but not at respective sampling locations (Table 12). In addition, a statistically significant, but weak relationship ($n = 52$, $r^2 = 0.13$, $F_{1, 50} = 7.422$, $p = 0.009$) was detected between monthly log transformed ridley CPUE and blue crab CPUE from entanglement nets (Table 12).

Table 10. Summary of least squares linear regression analysis for monthly and annual Kemp's ridley CPUE versus blue crab CPUE from trawl samples taken at and across study areas in the NW Gulf (1993-2002).

	Monthly CPUE [log (CPUE+1)]				Annual Mean CPUE			
	n	R^2	F	p	n	R^2	F	p
Sabine Pass	45	0.11	5.412	0.025	10	0.23	2.443	0.157
Calcasieu Pass	25	0.05	1.279	0.270	8	< 0.01	0.023	0.886
All Sites Combined	n/a	n/a	n/a	n/a	10	0.20	2.056	0.189
LDWF	n/a	n/a	n/a	n/a	8	-0.30	2.583	0.159

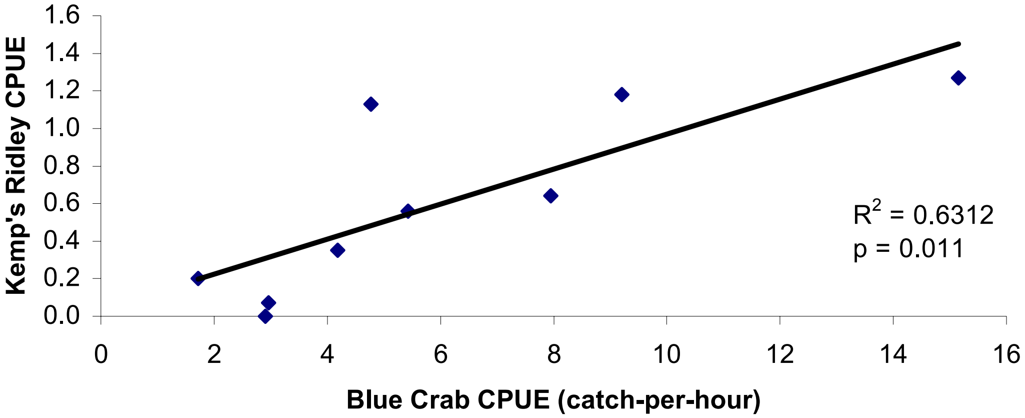


Figure 31. Annual mean Kemp's ridley CPUE versus blue crab CPUE from TPWD trawl samples collected near Sabine Pass, TX (west side CPUE only).

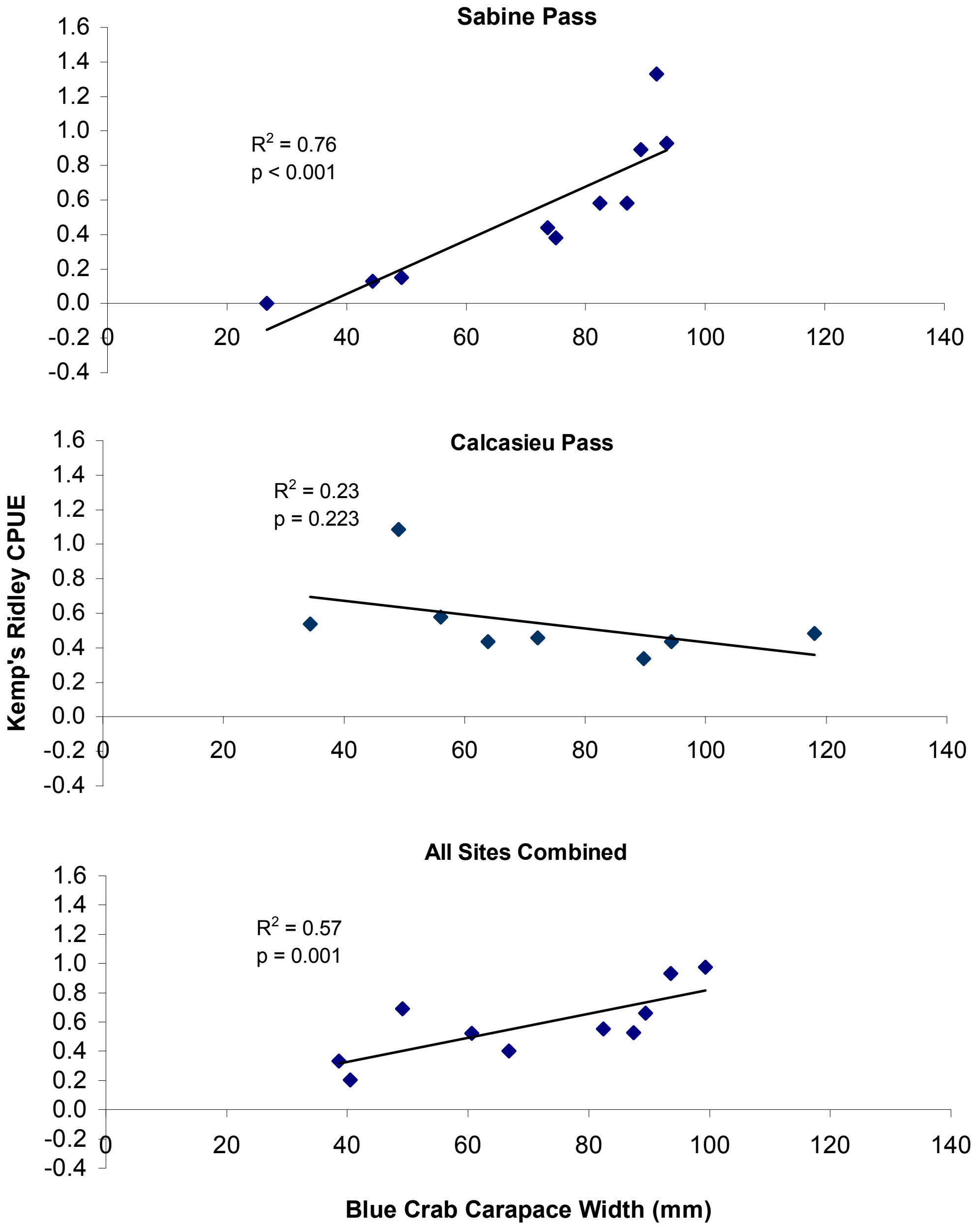


Figure 32. Annual mean Kemp's ridley CPUE versus mean blue crab carapace width at Sabine and Calcasieu Passes and all sites combined in the NW Gulf during 1993-2002.

Table 11. Multiple regression results for annual mean Kemp's ridley CPUE versus blue crab CPUE and size from trawl samples taken at Sabine Pass (1993-2002).

	Blue crab CPUE	Blue crab size *
n	10	10
slope	0.0058	0.0151
t	0.242	3.930
p	0.815	0.006*

* significant result at an alpha = 0.05

Table 12. Summary of least squares linear regression analysis for monthly and annual Kemp's ridley CPUE versus blue crab CPUE from entanglement nets at and across study areas in the NW Gulf (1993-2002).

	Monthly CPUE [log (CPUE+1)]				Annual Mean CPUE			
	n	**R^2**	**F**	**p**	**n**	**R^2**	**F**	**p**
Sabine Pass *	52	0.13	7.422	0.009*	10	0.02	2.040	0.191
Calcasieu Pass	34	0.02	0.345	0.564	8	< -0.01	0.200	0.674

Note: Annual mean blue crab CPUE across all sites combined was -1/x transformed to linearize the relationship between factors.

* significant result at an alpha = 0.05

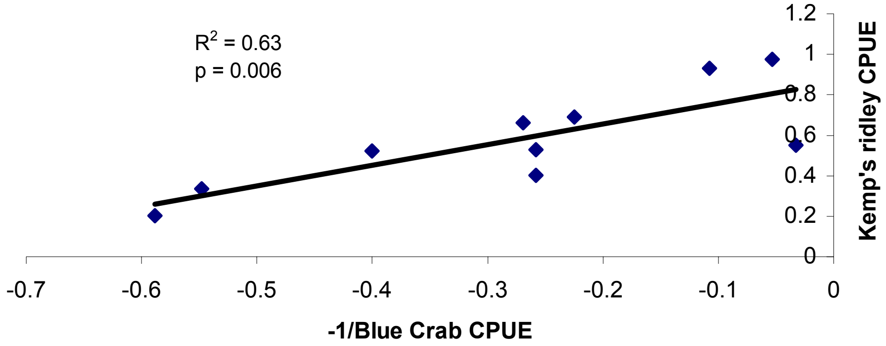


Figure 33. Annual mean Kemp's ridley CPUE versus transformed blue crab CPUE from entanglement nets (-1/x) over all sites combined in the NW Gulf during 1993-2002.

Kemp's Ridley Relationship to Shrimping Activity/By-catch

Analysis of Kemp's ridley abundance at stations west of Sabine Pass before, during and after the Texas Closure to shrimping yielded significant differences (ANOVA: n = 21, $F_{2,18} = 4.663$, p = 0.023), with post hoc comparisons revealing higher ridley CPUE during the closure period (mid-May to mid-July) ($\bar{x}$ = 1.23 ± 0.27/km-hr, n = 7) than after ($\bar{x}$ = 0.38 ± 0.13/km-hr, n = 7)(p = 0.007). However, there was no difference in ridley CPUE when comparing the period before the closure ($\bar{x}$ = 0.88 ± 0.17/km-hr, n = 7) to during the closure (p = 0.218). Furthermore, no significant difference existed between ridley CPUE (west-side only) at Sabine and Calcasieu Passes during the closure (Paired T-test: n = 7, $t_6 = -0.456$, p = 0.658). Ten of 32 sampling days in 2001-2002 yielded ridley captures with shimping activity in the area; 15 days had no captures but shrimp boats were present; 4 days had ridley captures but no shrimping activity; and 3 days had no ridleys and no shrimp boats in the area. The null hypothesis that ridley captures and shrimping activity are independent was not rejected (X^2 Test of Independence: $X^2_{1,\,0.05} = 0.672$, $p > 0.05$), and thus ridley occurrence was not significantly associated with shrimping activity on a localized scale.

Discussion

Blue crab abundance at NW Gulf study sites was highly variable between years and collection methods. Lack of correspondence between various measures of blue crab CPUE was most likely due to differences in gear selectivity and timing/location of sampling. Fishery-independent trawl surveys from TPWD and LDWF were conducted year-round and encompassed a larger area than the spatially and temporally restricted STFERL samples. Nonetheless, integration of different data sources into this assessment provided a more comprehensive characterization of the blue crab population in Kemp's ridley developmental habitat.

Abundance patterns from fishery-independent and fishery-dependent surveys suggest a high degree of inter-annual variability within blue crab stocks of the NW Gulf. Although other research has documented a positive link between blue crab productivity and freshwater inflow (More, 1969, Guillory, 2000), there was no clear, biologically meaningful relationship between the two in this study. In fact, the data suggested a negative relationship between abundance of larger entangled blue crabs and freshwater inflow to the Sabine-Neches estuary. This trend may be related to the sampling location and greater presence of gravid female crabs at this time on the beachfront. Increased freshwater inflow may result in reduced salinities that repel spawning females to open Gulf habitats. However, this was probably not the case because no significant negative correlation between inflow and salinity was detected. Any positive effect of freshwater inflow on blue crab abundance in nearshore Gulf waters might be seen geographically across the major Texas bay systems. Fishery-independent trawl surveys conducted in Gulf waters by TPWD indicate that highest blue crab CPUE is found offshore of the Sabine-Neches estuary (Fig. 34), which also receives the most freshwater (11.8 million cubic km/year optimum inflow) (TNRCC, 2002). Blue crab catch rates then tend to decrease south along the coast into more arid regions (i.e. near the Lower Laguna Madre). As such, nearshore habitat adjacent to positive estuaries on the upper Texas coast may provide greater foraging opportunities for juvenile ridleys compared to those in south Texas Gulf waters. Reports of higher abundance of juvenile ridleys along the upper Texas coast (Manzella and Williams, 1992; Cannon *et al.*, 1994; Landry and Costa, 1999) tend to support this conclusion.

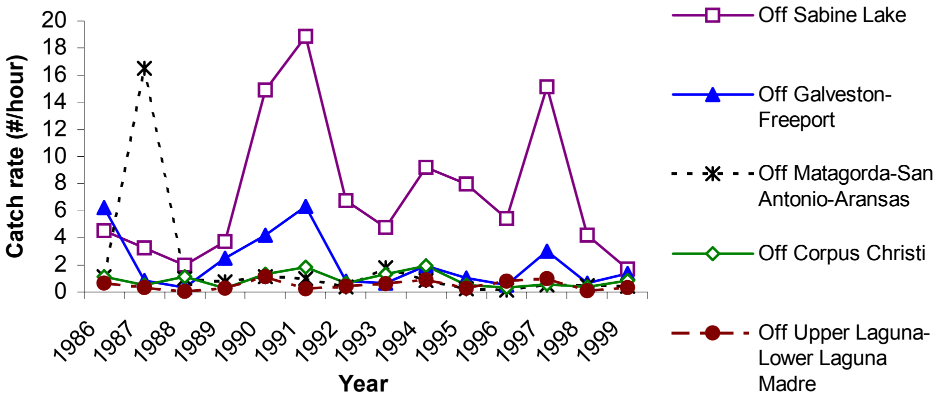


Figure 34. Blue crab catch rates from TPWD Gulf monitoring trawls in nearshore waters adjacent to major Texas estuaries (1986-1999).

There also was no indication from crab abundance trends that increasing expansion of summer hypoxia off the Louisiana coast concentrated crabs into shallower nearshore waters on an annual basis. The significantly greater trawl crab CPUE at sampling stations west of Sabine Pass may be attributed, in part, to water vortices created by long shore currents and prevailing southerly winds at this site. Sediment deposition within this entrained water mass produces soft muddy substrates characteristic of prime blue crab habitat (Perry and McIlwain, 1986; Britton and Morton, 1989). Thus, the propensity for blue crabs to inhabit such muddy, turbid areas probably explains the lack of a significant relationship with visibility. Because blue crabs have such a wide tolerance to environmental parameters, it is more likely that fluctuations in blue crab abundance are related to post-settlement (i.e. beyond the larval stages) biotic pressures, such as predation on juveniles, rather than direct effects from abiotic factors (Morgan *et al.*, 1996; Guillory, 2000).

Although there has been a 40% decline in Texas blue crab landings since 1987 (Wilson, 2001), similar trends were not reflected by TPWD fishery-independent trawl samples collected in Gulf waters near Sabine Pass for the same time period (1986-2001). This also was the case with other crab statistics at Sabine and Calcasieu Passes. Consequently, these measures seem to imply sufficient blue crabs are available to Kemp's ridleys foraging within nearshore waters of the NW Gulf. However, this does not take into consideration the complex issue of prey quality, particularly with regard to crab size. The 40% decline in Texas blue crab landings applies only to larger, legal size crabs (> 127 mm). Significant declines in mean carapace width were observed for blue crabs from all locations in the NW Gulf of Mexico, especially since 1997. A similar trend existed for crabs collected in TPWD surveys at Sabine Pass since 1986. This reduction in size corresponds with recent assessments of the blue crab stock and is symptomatic of "growth overfishing" (Guillory *et al.*, 1998; Hammerschmidt *et al.*, 1998). This may eventually have consequences on future blue crab population growth and stability, and could affect the primary prey base of Kemp's ridley sea turtles and other nearshore crab consumers.

The decline in blue crab size also may have reduced the attractiveness of Kemp's ridley foraging habitat in NW Gulf study areas, particularly at Sabine Pass. There was a strong relationship between ridley CPUE and blue crab size at Sabine Pass, with both variables showing concurrent declines since 1997. Even though ridley CPUE was somewhat related to crab abundance (i.e., a correspondence between greater ridley and trawl crab CPUE at stations west of Sabine Pass, as well as a significant relationship between ridley and TPWD trawl crab CPUE), blue crab size appeared important to explaining patterns of ridley abundance. It is not known if Kemp's ridley sea turtles prefer a certain crab size, but smaller crabs, even in high quantities, may not be desirable because they: 1) are more difficult to locate in the turbid nearshore environment; and/or 2) do not provide adequate sustenance for the energetic needs of *L. kempii*. Immature blue crabs (<127 mm) provide about 0.08 kcal/g live weight and thus a turtle would have to consume much higher quantities of small crabs to equal the energetic quality of larger/mature crabs (1.04 kcal/g live weight) (Thayer *et al.*, 1973). However, there also may be upper limits on the size of crab ridleys are able to capture and eat due to morphological constraints of body and mouth size. Most likely, there is a threshold above which blue crabs are more easily detected and/or an optimum size range of crabs that provides the most energetic return for the effort spent locating and capturing crab prey (Krebs and Davies, 1993). A lack of large crabs may result in an expansion of ridley foraging habitat in order to locate more suitable prey. This may explain the lower percentage of wild ridley recaptures in recent years. Reduced prey quality that results in ridleys traveling over greater distances while foraging could ultimately impact the energy available for ridley growth, maintenance and reproduction.

Conclusive statements regarding the effect of prey size and abundance on ridley occurrence are tempered by inconsistent results across sample locations. There was no significant difference in crab size (or abundance) between sampling locations that might explain the shift in ridley abundance from Sabine Pass to Calcasieu Pass after 1997.

Additionally, there was no significant relationship between ridley CPUE and crab size at Calcasieu Pass, even though a decline in size was observed. The lack of a statistically significant relationship at Calcasieu Pass may be an artifact from this location, which was less consistently sampled than that at Sabine Pass. Nevertheless, more research is needed to ascertain the role of prey size on ridley abundance in developmental foraging habitats.

Shrimping activity had no noticeable effect on ridley CPUE in general or on similar statistics calculated before, during and after the Texas Closure (mid-May to mid-July). Any differences between CPUE statistics, such as the lower CPUE observed after the closure, were probably related more to monthly differences in ridley abundance than to a shrimping effect (i.e., higher CPUE was observed during April and May compared to August-October). Also, because shrimping effort in this region is high (Sub-Area 17/NW Gulf) (Nance, 1993), there was a greater chance of sighting a shrimp boat compared to the relatively rare event of a ridley capture. Although this examination of shimping activity/bycatch as an attractant to ridley foraging was inconclusive, potential interaction between ridleys and shrimping activity in nearshore developmental habitat remains high (Epperly *et al.*, 2002). Further research may be necessary to ascertain the role of shrimping bycatch in the Kemp's ridley diet, and how it may affect ridley foraging ecology, habitat use, and distribution patterns.

This chapter provides evidence that prey availability is a significant factor influencing ridley occurrence in nearshore waters of the NW Gulf. However, there are still many aspects of juvenile foraging ecology that are unknown, such as prey size preference, feeding rate, energetic requirements and mechanisms for locating prey, and future research should address these gaps in our knowledge. It is important that we gather more information on feeding behavior and quality of the blue crab resource because of potential impacts to ridley distribution, growth, maturity and overall fitness.

CHAPTER V

PREDATION PRESSURE

Introduction

Presence of predators may influence habitat use and movements of Kemp's ridley sea turtles, and thus, is included as a major component in the conceptual model of ridley occurrence presented in Chapter I (Fig. 2, Table 1). The specific portion of the model that pertains to this influence is isolated in Fig. 35. Although the risk of in-water predation declines as sea turtles grow, missing flippers and bite scars found on juvenile, subadult, and adult individuals are evidence of vulnerability to predation by sharks (Márquez, 1994). Because animals must balance between acquiring necessary food resources and avoiding predation, the decision to forage in the presence of predators is often based on criteria such as: 1) energetic needs of the forager (i.e. hunger state); 2) abundance of predators (i.e. predator-prey encounter rate); 3) behavioral and morphological adaptations to avoid or escape injury; and 4) environmental aspects that may provide shelter or refuge from predators (Lima and Dill, 1990; Krebs and Davies, 1993). These considerations imply that Kemp's ridleys may be deterred from foraging in developmental habitats occupied by sharks, even if prey (blue crabs) are plentiful.

The bull shark (*Carcharhinus leucas*) is the most likely shark predator of ridleys in the nearshore waters of the NW Gulf (Shipley, 2000). Clark and von Schmidt (1965) documented *C. leucas* as the most commonly encountered shark in waters less than 14 m deep along the central Gulf coast of Florida, while others have reported this species' presence in shallow waters of the north-central Gulf westward to Texas (Caillouet *et al.*,

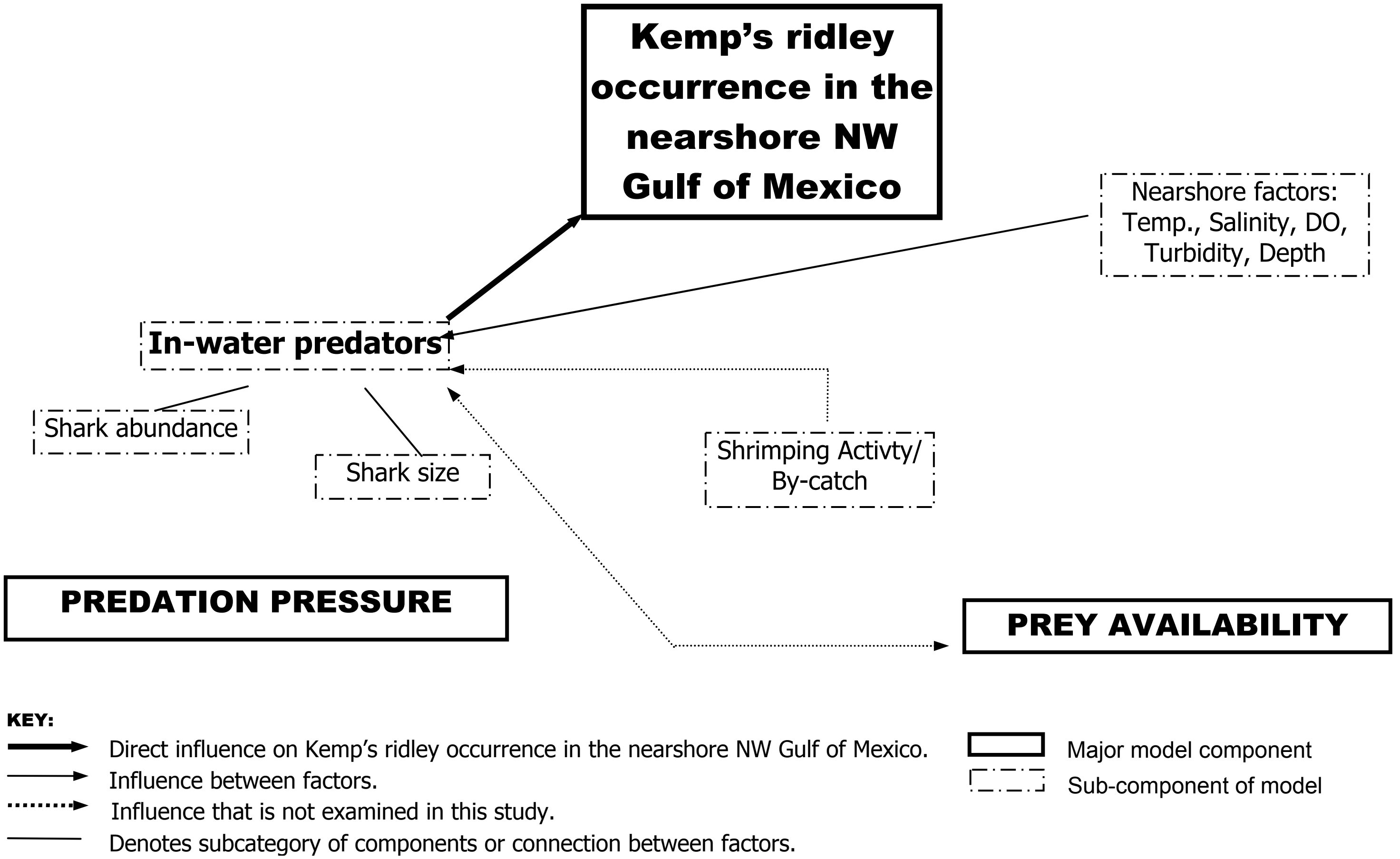


Figure 35. Portion of conceptual model detailing the hypothesized influence of predation pressure on Kemp's ridley occurrence in nearshore waters of the NW Gulf of Mexico.

1969; Branstetter, 1981; Castro, 1983; Compagno, 1994; Grace and Henwood, 1997; Shipley, 2000). The bull shark is a heavy-bodied species, most frequently associated with estuaries and adjacent marine waters (Castro, 1983), but also capable of penetrating freshwater/low salinity areas (Thomerson *et al.*, 1977).

Bull sharks are aggressive, opportunistic feeders with a varied diet that includes sea turtles, but mostly consists of fish (e.g. other sharks, stingrays, mullet, catfish, jacks, menhaden) and, to a lesser extent, crustaceans (e.g. crabs and shrimp), porpoises, and mollusks (Clark and von Schmidt, 1965; Branstetter, 1981; Compagno, 1984; Snelson *et al.*, 1984). Adult female bull sharks use nearshore waters during April-June for pupping and juvenile bull sharks use these waters as an important nursery ground (Clark and von Schmidt, 1965; Branstetter and Stiles, 1987). Shipley (2000) found that 74% of bull sharks captured in entanglement nets near Sabine Pass, TX were juveniles between 111-225 cm total length (TL). It is presumed that these sharks are large enough to inflict damage on 20-40 cm SCL juvenile ridleys, the dominant life history stage found in developmental habitat at Sabine and Calcasieu Passes. Conversely, bull sharks primarily use senses other than vision to locate and capture prey. As such, attacks on ridleys may be more opportunistic and/or a result of the bull shark's documented aggressive behavior (Castro, 1983; Compagno, 1984).

This chapter takes a first look at possible predation pressure on juvenile Kemp's ridleys by bull sharks in nearshore waters of Sabine and Calcasieu Passes, by characterizing the abundance and distribution of bull sharks in the study area and comparing it to CPUE on ridleys. I hypothesized that there would be a negative relationship between Kemp's ridley and shark CPUE. Such a pattern would support the notion that the turtles avoid bull sharks. The frequency of probable shark-inflicted injuries on captured ridleys also was quantified in order to make inferences about predator-prey interactions (Schoener, 1979, Heithaus *et al.*, 2002). A relatively high frequency of shark-inflicted wounds to Kemp's ridleys is hypothesized to infer high encounter rate and low ability of ridleys to avoid or escape attack.

Materials and Methods

Study areas and Kemp's ridley in-water survey methods are described in Chapter II. Bull sharks captured in entanglement nets concurrent to ridley sampling (April-October, 1993-2002) were enumerated, and CPUE was measured in the same fashion as *L. kempii* (number/km-hour). Water temperature (°C), salinity (ppt), visibility (m) and dissolved oxygen content (mg/L) were obtained via procedures presented in Chapter III and IV. The frequency of ridleys with shark-inflicted injuries was assessed via a photographic survey of all captured turtles (n = 600). Shark-inflicted injuries were identified using criteria described by Heithaus *et al.* (2002) that include: at least 1/3 of a flipper missing; carapace with a crescent-shaped portion removed that could only have been caused by a shark; and/or presence of obvious tooth marks or puncture wounds (Fig. 36). Although identification of possible shark-inflicted injuries followed these criteria, individual determinations were somewhat subjective. Also, it was virtually impossible to determine the geographical location where injuries were inflicted as some may have occurred outside the study area.

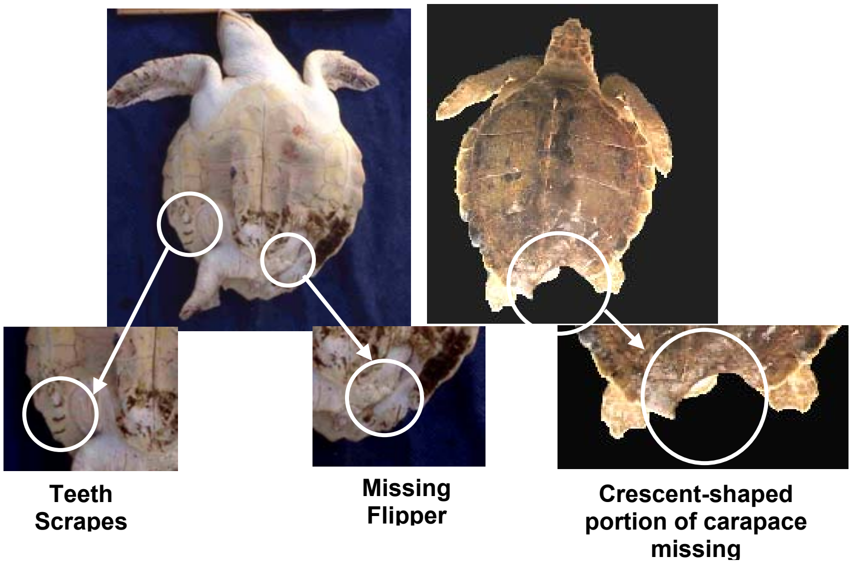


Figure 36. Examples of possible shark-inflicted injuries to Kemp's rideys captured at Sabine and Calcasieu Passes.

Data Analysis

Monthly ridley CPUE values were log transformed [Log (CPUE+1)] to approximate a normal distribution. One-way ANOVA was used to identify differences in ridley CPUE and size between years (across all sites combined and at Sabine and Calcasieu Passes), while a t-test was used to examine differences in these parameters between study areas (Sabine and Calcasieu Passes). ANOVA also was used to detect differences in bull shark CPUE between sample months (April-October) and netting stations (1, 3, 4, 5) across all study sites combined. Post hoc comparisons were conducted using the Least Significant Difference (LSD) test. Sampling biases described in Chapter II also were taken into consideration when interpreting differences in monthly shark CPUE between netting stations. Trends in annual mean bull shark abundance were assessed via the same method described in previous chapters (null hypothesis: slope of the regression line = 0). Least squares linear regression was used to examine the relationship between bull shark CPUE and abiotic factors (water temperature, salinity, visibility, and DO) on a monthly and annual basis. The relationships between monthly and annual Kemp's ridley CPUE versus bull shark CPUE also were assessed using least squares linear regression. Frequency of shark-inflicted injury to Kemp's ridleys was measured as the percentage of ridleys exhibiting wounds out of the total number captured. A t-test was used to determine differences in frequency of shark-inflicted injuries between Sabine and Calcasieu Passes for comparable years. Mean values in this chapter are expressed as $\bar{x}$ ± standard error. All statistical analyses were conducted using an α level of 0.05 in Microsoft Excel and SPSS statistical software packages.

Results

Bull Shark Abundance and Distribution

Monthly bull shark CPUE in nearshore waters of the NW Gulf of Mexico (1993-2002) was highly variable, with no captures over several months and others with relatively high capture rates (Table 13). Monthly CPUE for bull sharks across all study areas was not significantly different (ANOVA: $n = 86$, $F_{6, 76} = 1.602$, $p = 0.158$), although slightly higher abundance was observed during April-August compared to that in September and October (Fig. 37).

Table 13. Monthly bull shark CPUE at Sabine and Calcasieu Passes and annual CPUE statistics for all sites combined in the NW Gulf during 1993-2002.

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Sabine Pass										
April	0.67	0.00	0.00	0.00	32.50					
May	3.09	0.00	1.01	0.75	2.01	2.98	3.68	0.37	4.13	
June	0.25	5.63	0.00	2.16	1.73	0.00	0.44	0.34	2.99	1.04
July	2.13	3.89	0.48	0.88	4.52	0.42	6.86	0.68	0.81	4.98
August	2.34	0.00	1.78	3.58	2.95	0.22	1.61	2.70	1.14	
September	1.71	1.04	0.00	0.00	0.00					
October	0.00	0.79	0.00	0.09						
n	166	92	41	72	244	88	173	34	79	55
Mean	**1.46**	**1.62**	**0.47**	**1.07**	**7.28**	**0.91**	**3.15**	**1.02**	**2.27**	**3.01**
STD	1.17	2.24	0.70	1.34	12.44	1.40	2.81	1.13	1.57	2.78
STERR	0.44	0.85	0.26	0.51	4.70	0.53	1.06	0.43	0.59	1.05
CV	80.09	138.30	148.77	126.22	170.80	153.92	89.41	0.00	69.15	92.36
Calcasieu Pass										
May	0.00						0.00	5.44	0.40	0.21
June	0.24	33.84	0.00			6.07	1.87	0.20	0.08	0.32
July	0.48	2.01	0.20			1.25	0.17	0.00	0.82	0.24
August		0.10	0.00			0.00	0.17	0.00	0.13	
September	0.00	0.00					0.00			
October	0.00	0.00	0.00				0.00			
n	4	113	2			62	23	44	13	13
Mean	**0.14**	**7.19**	**0.05**			**2.44**	**0.37**	**1.41**	**0.36**	**0.26**
STD	0.21	14.92	0.10			3.21	0.74	2.69	0.34	0.06
STERR	0.10	6.67	0.04			1.43	0.33	1.20	0.15	0.03
CV	149.67	207.55	200.00			131.48	200.26	190.53	94.60	23.22
All Sites Combined										
n	170	205	43	134	267	132	186	47	92	68
Mean	**0.91**	**3.94**	**0.32**	**1.07**	**7.28**	**1.56**	**1.48**	**1.22**	**1.31**	**1.36**
STD	1.10	9.59	0.58	1.34	12.44	2.25	2.24	1.92	1.46	2.05
STERR	0.32	2.77	0.18	0.51	5.08	0.85	0.75	0.68	0.52	0.92
CV	121.27	243.32	184.14	126.22	170.80	144.07	151.10	157.91	111.51	151.08

Blank = Not Sampled

CPUE shown for Calcasieu 1999 and 2000 also includes that from Mermentau Pass, LA.

n = total number of individuals captured; STD = Standard Deviation; STERR = Standard Error; CV = Coefficient of Variation

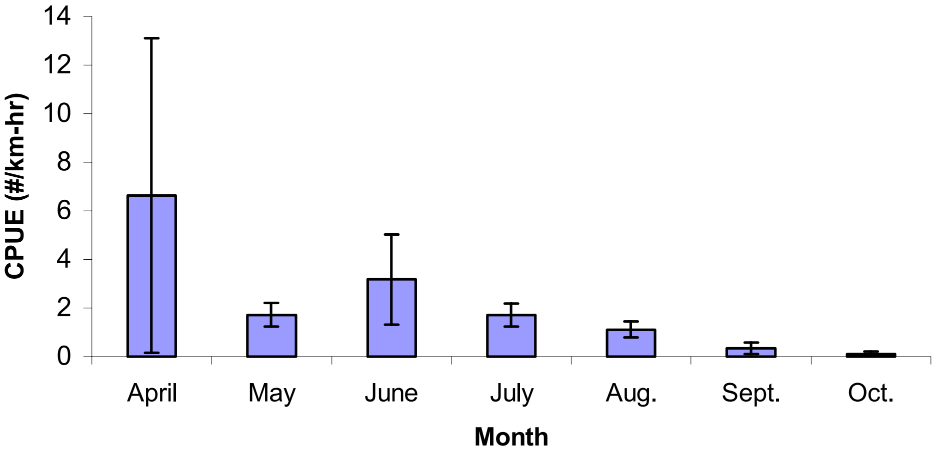
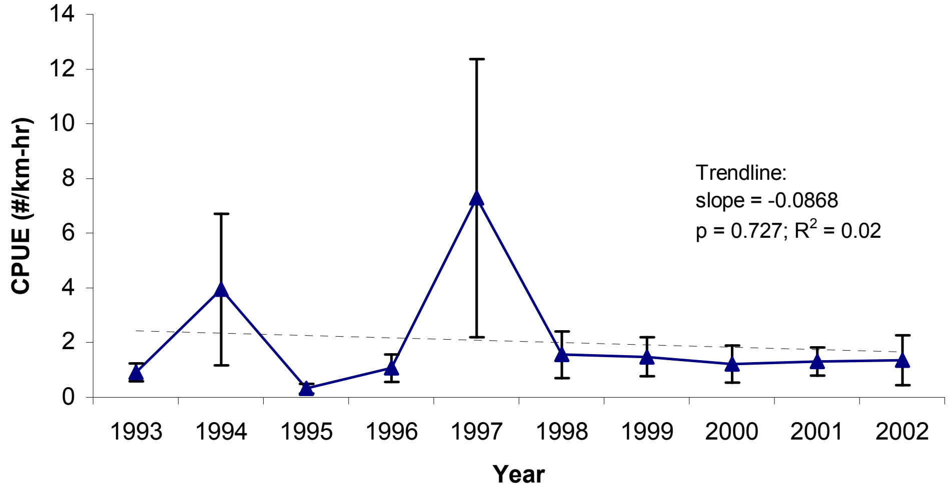


Figure 37. Mean bull shark CPUE (w/ standard error bars) for months entanglement netting occurred over all sites in the NW Gulf during 1993-2002.

Annual mean bull shark abundance across all NW Gulf locations varied among years, with peaks in CPUE in 1994 (3.94 ± 2.77/ km-hr, n = 12) and 1997 (7.28 ± 5.08/ km-hr, n = 6) (Fig. 38a), but the variation was not statistically significant (ANOVA: n = 86, $F_{9, 76}$ = 1.288, p = 0.259) and exhibited no particular trend (n = 10, slope = -0.0868, r^2 = 0.02, F_8 = 0.727, p = 0.727). There also was no statistically significant trends in annual mean CPUE at study areas (Sabine Pass: n = 10, slope = 0.1283, r^2 = 0.04, F_8 = 0.320, p = 0.589; Calcasieu Pass: n = 8, slope =-0.2677, r^2 = 0.14, F_6 = 0.960, p = 0.365) or significant variation among years (ANOVA: Sabine Pass n = 52, $F_{9, 42}$ = 1.579, p = 0.153; Calcasieu Pass n = 34, $F_{7, 26}$ = 0.895, p = 0.525) (Fig. 38b). Significantly greater bull shark abundance was observed at Sabine Pass during the entire study period ($\bar{x}$ = 2.14 ± 0.64, n = 52) (t-test: n = 86, t_{84} = 2.431, p = 0.017) and during 1998-2002 (n =

a) **All Sites Combined**



b) **Sabine and Calcasieu Passes**

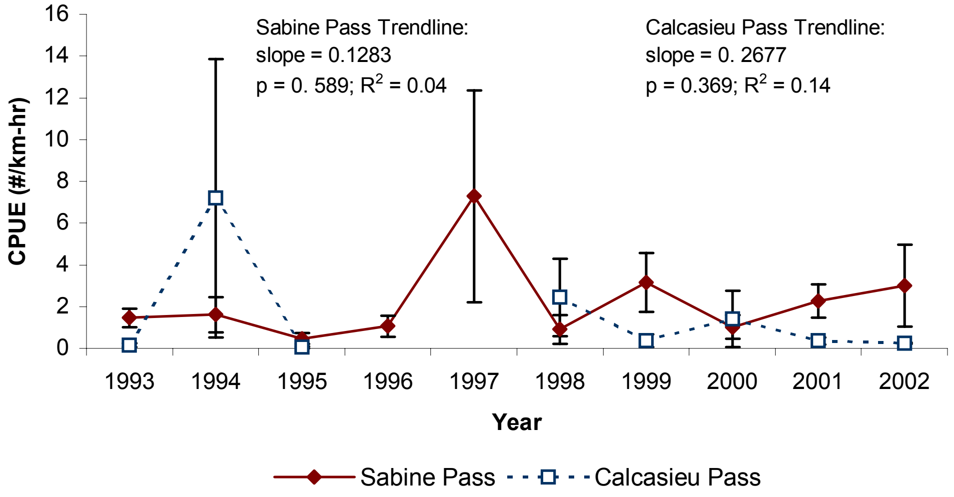


Figure 38. Annual mean bull shark CPUE (w/ standard error bars) across all sites combined (a) and at Sabine and Calcasieu Passes (b) during 1993-2002.

38, t_{36} = 2.530, p = 0.016). However, no statistical difference was detected between the passes for years 1993-1995 (n = 35, t_{33} = 0.736, p = 0.467). A comparison of shark CPUE across all study areas combined was significantly different among netting stations (ANOVA: n = 177, $F_{3,173}$ = 4.108, p = 0.008). LSD post hoc comparisons revealed considerably higher levels at the west beachfront station (station #3) than at either jetty station (stations #1 or 4) (p = 0.002 and 0.009, respectively). No differences were detected between jetty stations 1 and 4 (p = 0.880) or beachfront stations 1 and 3 (p = 0.249)

No significant relationships were found between shark abundance and abiotic factors (water temperature, salinity, visibility and dissolved oxygen content) on a monthly or annual basis (Table 14).

Relationship Between Kemp's Ridley and Bull Shark CPUE

A statistically significant positive relationship was detected between annual mean Kemp's ridley and bull shark CPUE (n = 10, r^2 = 0.48, $F_{1,8}$ = 7.502, p = 0.025) (Fig. 39) across all sample sites in the NW Gulf, with concurrent abundance peaks observed in 1994 and 1997. However, there were no significant relationships between Kemp's ridley and shark CPUE at individual study areas by month or year (Table 15).

Table 14. Summary of least squares linear regression results for monthly and annual bull shark CPUE versus abiotic factors at and across study areas in the NW Gulf (1993-2002).

	Monthly Mean Abiotic & [Log (shark CPUE+1)]				Annual Means			
	n	R^2	F	p	n	R^2	F	p
Sabine Pass								
Water temperature	52	< 0.01	0.058	0.811	10	< -0.01	0.051	0.827
Salinity	52	< 0.01	0.010	0.920	10	< 0.01	0.040	0.846
Water visibility	52	-0.05	2.545	0.117	10	< 0.01	0.001	0.975
Dissolved oxygen	31	< -0.01	0.073	0.788	7	-0.05	0.233	0.649
Calcasieu Pass								
Water temperature	34	0.06	1.948	0.172	8	0.07	0.446	0.529
Salinity	34	-0.10	3.471	0.072	8	< 0.01	0.023	0.885
Water visibility	34	-0.09	3.095	0.088	8	0.05	0.309	0.598
Dissolved oxygen	20	0.05	0.884	0.359	5	-0.26	1.054	0.380
All Sites Combined								
Water temperature	n/a	n/a	n/a	n/a	10	< 0.01	0.063	0.808
Salinity	n/a	n/a	n/a	n/a	10	< -0.01	0.002	0.969
Water visibility	n/a	n/a	n/a	n/a	10	0.06	0.483	0.507
Dissolved oxygen	n/a	n/a	n/a	n/a	7	< 0.01	0.013	0.844

Note: Sabine Pass DO statistics are for 1996-2002

Calcasieu Pass DO statistics are for 1998-2002

Sabine Pass Freshwater Inflow statistics are for 1993-1999

* signifcant result at an alpha = 0.05

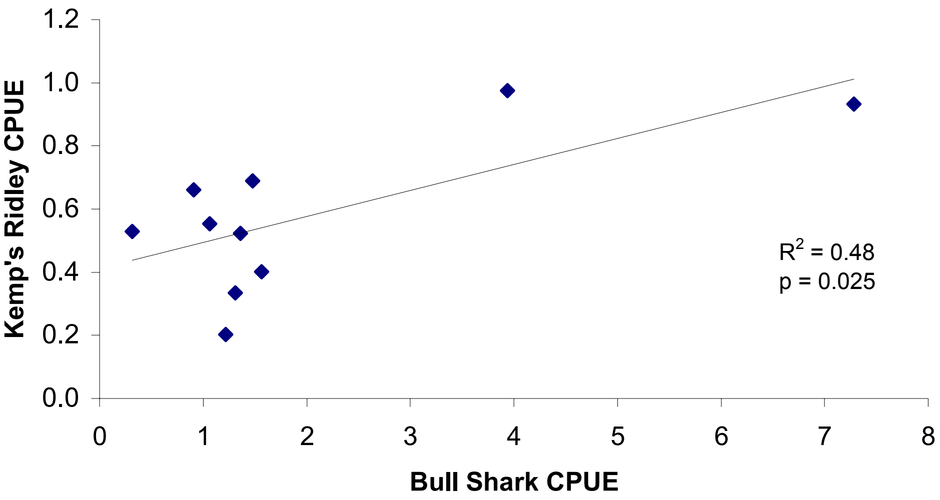


Figure 39. Annual mean Kemp's ridley CPUE versus bull shark CPUE for all sites combined in the NW Gulf during 1993-2002.

Table 15. Summary of least squares linear regression analysis for monthly and annual Kemp's ridley CPUE versus bull shark CPUE at study areas in the NW Gulf (1993-2002).

	Monthly CPUE [log (CPUE+1)]				**Annual Mean CPUE**			
	n	**R^2**	**F**	**p**	**n**	**R^2**	**F**	**p**
Sabine Pass	52	0.02	0.914	0.344	10	0.04	0.370	0.560
Calcasieu Pass	34	< 0.01	0.011	0.917	8	-0.03	0.161	0.702

* significant result at an alpha = 0.05

Frequency of Shark-Inflicted Injuries

The percentage of Kemp's ridleys with shark-inflicted injuries across all sites fluctuated between 0 and 15.2% during 1993-2002, but averaged about 6.0% ± 1.3% (n

= 10) over all years combined (Fig. 40). Although the highest frequency of injuries was observed in 2002 (15.2%), there was no significant trend across years (n = 10, slope = -0.0034, $r^2 = 0.06$, F_8, 0.490, p = 0.504). There was a slightly higher, but statistically non-significant mean frequency of injured ridleys at Calcasieu Pass (6.9% ± 2.6%, n = 8) than at Sabine Pass (4.8% ± 2.0%, n = 8) (t-test: n = 16, $t_{14} = 2.145$, p = 0.509). There was no relationship between frequency of shark-inflicted injuries and both Kemp's ridley and bull shark abundance patterns (Multiple regression: n = 10, $r^2 = 0.08$, $F_{2,7} = 0.313$, p = 0.741).

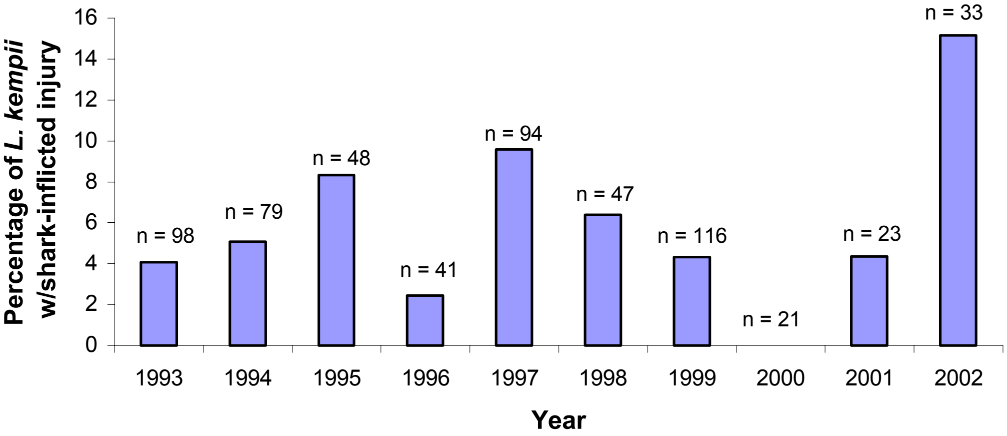


Figure 40. Percentage of *L. kempii* exhibiting shark-inflicted bite injuries at all sites combined in the NW Gulf during 1993-2002. (n = total number of ridleys captured; Overall mean injury frequency = 6.0 ± 1.3%, n = 10)

Discussion

Prevalence of bull sharks in nearshore NW Gulf waters during April to August coincides with other accounts of this species' seasonal occurrence, especially with regard to juvenile abundance (Clark and von Schmidt, 1965; Castro, 1983). However, lack of year-round sampling in this study prevented comparison across all months/seasons. Failure to detect significant relationships between bull shark CPUE and environmental parameters suggests no influence of these factors on shark abundance during the time period sampled. Numerous prey items, including catfish (*Arius felis* and *Bagre marinus*), Gulf menhaden (*Brevoortia patronus*), and various ray species (*Dasyatis americana*, *D. sabina*, *D. centroura*, and *Rhinopterus bonasus*) were commonly encountered at sampling locations, and may have influenced bull shark abundance and distribution (Clark and von Schmidt, 1965, Branstetter, 1981, Compagno, 1984, Snelson *et al.*, 1984).

Kemp's ridleys foraging in nearshore waters may minimize the probability of their being bitten or killed by bull sharks via two main mechanisms: 1) avoiding shark-inhabited areas and 2) evasive maneuvers (Heithaus *et al.*, 2002). Abundance patterns at netting stations and across sampling years provided no strong indication that Kemp's ridleys avoid bull sharks. Although bull shark CPUE was higher at beachfront locations, there was no difference in ridley occurrence at jetty and beachfront sites that would imply an aversion to shark-prone areas. In fact, peaks in annual mean ridley CPUE coincided with elevated shark abundance in 1994 and 1997.

Both juvenile ridleys and bull sharks aggregate in nearshore waters during summer months to feed on seasonally abundant prey. The peaks in CPUE observed in 1994 and 1997 may have been related to greater prey availability during these years.

However, this simultaneous use of nearshore waters also may have increased the likelihood of encounters between ridleys and bull sharks.

The ability of sea turtles to avoid or escape shark attack may be related to swimming velocity, maneuverability and evasive measures (Heithaus *et al.*, 2002). There are no data on the speed of ridleys when avoiding capture, but tracking studies have shown a range of 0.02-11.0 km/hr and an overall mean of 1.0 ± 0.05 km/hr during normal activity (Renaud, 1995). This velocity is probably not fast enough to elude a pursuing bull shark, and, as such, additional mechanisms may be employed to circumvent capture. The defensive behavior of ridleys may be similar to that of hawksbills who position their carapace perpendicular to the direction of attack, thus making it more difficult for a shark to grasp the flat surface (Witzell, 1983; Márquez, 1994).

It is possible that the juvenile bull sharks encountered in the study area do not pose a significant threat to foraging ridleys. The average percentage of Kemp's ridleys possessing shark-inflicted injuries (6%) is similar to the result reported by Heithaus *et al.* (2002) for green turtles (~ 5%) in Shark Bay, Western Australia, which was considered low in comparison to male loggerhead injury frequency (~ 60%) from the same location. The low injury frequency for Kemp's ridleys in this study, coupled with the lack of a negative relationship between ridley and shark abundance, might suggest a low predation risk or encounter rate. However, it should be noted that injury frequency may not accurately reflect predation pressure because it does not include turtles that died as a result of shark predation.

Overall, Kemp’s ridleys were not deterred from using areas frequented by bull sharks. This result also might suggest that a ridley’s need to forage supersedes any threat of predation posed by bull sharks in the area. In addition, there were no differences in bull shark CPUE between study locations and sample years that might explain the shift in ridley abundance from Sabine Pass to Calcaseiu Pass after 1997. However, there are still many gaps in our understanding of shark predation on sea turtles. More information is needed on the frequency of predator-prey interactions and escape abilities of ridleys to ascertain whether in-water predators significantly impact juvenile ridley distribution, survival, and foraging success.

CHAPTER VI

SUMMARY AND CONCLUSIONS

Summary

This study utilizes 10 years of in-water survey data to characterize long-term abundance and distribution of Kemp's ridley sea turtles (*Lepidochelys kempii*) in developmental feeding grounds of the northwestern Gulf of Mexico. Additionally, a conceptual model is developed to assess factors hypothesized to influence Kemp's ridley occurrence in this region. Major model components are based on known aspects of Kemp's ridley biology and include nesting dynamics, environmental conditions, prey availability, and predation pressure. Information gathered by the present study is designed to aid in the management and continued recovery of this endangered species by increasing our knowledge of in-water life history stages and their habitat requirements.

Kemp's ridley nearshore occurrence was assessed via monthly entanglement operations conducted primarily at Sabine Pass, TX and Calcasieu Pass, LA during April-October 1993-2002. Although sampling was inconsistent among months and selection of netting stations was somewhat biased, analysis of annual trends and comparison of parameters among years and between study areas was possible.

Size distributions of *L. kempii* captured during this study were comprised primarily of post-pelagic juveniles through subadult life history stages (20-40 cm SCL). This trend is consistent with other reports of ridleys inhabiting shallow coastal benthic habitat (Ogren, 1989; Manzella and Williams, 1992; Schmid, 1995; Landry and Costa, 1999; TEWG, 2000). Slightly higher mean SCL was observed for ridleys at Calcasieu

Pass due to large subadults and adult females foraging in nearshore waters off the Louisiana coast.

Kemp's ridley annual mean CPUE across all study sites peaked in 1994, 1997, 1999 and 2002, thereby suggesting a 2-3 year cycle in abundance. This abundance pattern was significantly related to hatch success (with a 2-year lag) at the Rancho Nuevo nesting beach. However, there was no significant relationship between Kemp's ridley CPUE in nearshore waters and number of hatchlings leaving the nesting beach. In fact, juvenile ridley CPUE at netting sites remained relatively constant or decreased slightly, as number of hatchlings released from Rancho Nuevo continued to increase exponentially.

Strandings can provide an additional indicator of Kemp's ridley abundance and distribution over a larger geographic scale, if major factors affecting stranding rates, such as commercial shrimping effort, currents, and observer effort, are taken into consideration (Henwood and Stunz, 1987, Magnuson *et al.*, 1990; Caillouet *et al.*, 1991, Caillouet *et al.*, 1996). Shrimping effort within NMFS statistical sub-areas along the US coast exhibited no particular trends post-1990, and, thus, ridley stranding statistics from these regions may reflect nearshore abundance and distribution patterns (assuming observer effort and currents have remained relatively constant during the same time period). Declines in Texas ridley strandings since 1994, coupled with increasing Florida strandings since 1995, suggest a shift in distribution from the western to eastern Gulf in recent years. However, in-water capture data from Florida studies were insufficient to corroborate this explanation.

Kemp's ridley annual mean CPUE was most significantly related to blue crab size at Sabine Pass. However, a similar result was not observed at Calcasieu Pass. Furthermore, there were no differences in prey quantity or quality that would explain the shift in higher ridley CPUE from Sabine Pass to Calcasieu Pass after 1997. There also were no indications that ridley captures were significantly related to presence of shrimping activity/by-catch at study locations. Coastal waters of the NW Gulf serve as developmental feeding grounds for both juvenile ridleys and bull sharks, yet elevated bull shark abundance at netting sites did not appear to deter ridleys from utilizing nearshore habitat.

Conclusions

This study provides valuable information on Kemp's ridley long-term in-water dynamics that is essential for understanding this species' ecology and enhancing its management and recovery. Conclusions regarding the research hypotheses presented in Chapter I are summarized in Table 16.

Nesting dynamics and prey availability were conceptual model components appearing to have the greatest influence on nearshore ridley occurrence. The 2-3 year cycle in nearshore ridley abundance seems to be related to temporal patterns in clutch size or hatch success at Rancho Nuevo resulting from variability in nesting female fecundity and the remigration interval. Tag returns indicate 20% of adult females return to nest every year, 60% every 2 years, 15% every 3 years and 5% every 4 years (TEWG,

Table 16. Summary and evaluation of the research hypotheses presented in Chapter I.

Hypothesis	Evaluation
1) Kemp's ridley abundance at study areas will be significantly correlated with number of hatchlings released from Rancho Nuevo and patterns in nesting activity.	- Reject with regard to number of hatchlings - Accept with regard to hatch success
2) Kemp's ridley occurrence at study areas will be positively correlated with water temperature and salinity.	- Reject/ Inconclusive
3) Kemp's ridley occurrence at study areas will be associated with the abundance and size of blue crab prey.	- Accept with regard to blue crab size at Sabine Pass. - Inconclusive with regard to blue crab CPUE - Inconsistent results among parameters and study areas
4) Kemp's ridley occurrence at study areas will be negatively correlated with bull shark abundance and distribution.	Reject

2000). Neophyte nesters are typically less fecund than older turtles, producing smaller clutches with lower hatch success. Thus, peaks in clutch size and hatch success every

2-3 years may be related to the return of older, more fecund, nesting females. The significant relationship between ridley CPUE and hatch success (with a 2-year lag) also may be linked to greater hatchling or post-hatchling cohort survival, producing 2-3 year pulses in nearshore recruitment. However, the disparity between Kemp's ridley abundance in the NW Gulf and the exponentially increasing number of hatchlings leaving Rancho Nuevo raises the question, where are these progeny going? Assuming post-hatchling mortality rates did not drastically increase during the monitoring period, it is suspected that juvenile ridleys are recruiting to coastal locations outside of the NW Gulf study areas. Changes in ridley occurrence from the western to eastern Gulf (based on stranding statistics) may be related to fluctuations in circulation patterns that impact transport of this species from the pelagic realm to coastal waters, but no attempt was made to assess such a relationship in this study.

The decline in ridley CPUE observed at Sabine Pass since 1997 appears related to a concurrent reduction in blue crab size. This decrease in crab size corresponds to recent assessments of the Texas blue crab stock as symptomatic of "growth overfishing" (Hammerschmidt *et al.*, 1998). Excessive harvest of large crabs and subsequent juvenescence of the crab population may have resulted in sub-standard prey availability for juvenile ridleys foraging at Sabine Pass. Smaller crabs, even in high quantities, may less desirable because of the difficulty in locating them or their inability to provide adequate sustenance for energetic needs of *L. kempii*. Reduced prey quality may force ridleys to expand their foraging habitat to locate more suitable crabs. This also may explain the recent declines in ridley recaptures. However, conclusive statements regarding the relationship between Kemp's ridley abundance and blue crab size are tempered by a lack of corresponding evidence at Calcasieu Pass. Nonetheless, changes in prey quality may ultimately affect juvenile ridley growth, age at sexual maturity, and overall fitness (Bjorndal, 1997). It is concluded that subsequent studies should focus on understanding the energy requirements and feeding habits of Kemp's ridleys, as well as potential threats to their foraging success.

Lack of a negative relationship between Kemp's ridley and bull shark CPUE suggests, for ridleys, that the need to acquire food items outweighs the predation risk. The relatively low frequency of probable shark-inflicted injuries also suggests that juvenile bull sharks are not specifically targeting ridleys as prey and that attacks are most likely random events (i.e. "wrong place, wrong time" for injured ridleys). Thus, the threat of predation is not significant enough to elicit changes in habitat use.

Although this 10-year study generated the most long-term in-water dataset on Kemp's ridleys from the western Gulf of Mexico, funding and logistic limitations compromised its utility to produce definite conclusions about ridley dynamics in developmental habitat. Spatially- and temporally-restricted aspects of sampling rendered results from this study pertinent only to ridley occurrence at Sabine and Calcasieu Passes, and not necessarily representative of this species' abundance and distribution throughout the NW Gulf or other regions. Nonetheless, this study represents a more comprehensive approach to understanding Kemp's ridley in-water dynamics and provides information on ridley abundance patterns (2-3 yr cycle) and habitat associations. Additionally, this research has generated questions about the link between ridley nearshore abundance patterns and nesting success, the distribution of ridleys between the western and eastern Gulf, and role of prey quality (i.e. blue crab size) on foraging habitat selection. Future studies should address these questions by using the conceptual model presented herein as a framework for further analyses.

Increased nesting success at Rancho Nuevo in recent years is a promising sign of Kemp's ridley population growth, but without in-water abundance data, evaluation of recovery efforts is incomplete. The fact that sea turtles spend over 99% of their life at sea mandates that monitoring all life history stages be a management priority. Logistical difficulties and expense of in-water sampling should not be a barrier to gathering additional information on ridleys in nearshore developmental habitat, especially in light of the reproductive potential these individuals represent to the breeding population (Crouse *et al.*, 1987).

Future Research Recommendations

The following research recommendations are rendered to improve our understanding of Kemp's ridley in-water dynamics and essential habitat requirements:

- Marine resource management agencies such as NOAA/NMFS and other environmental entities provide funding prerequisite to a comprehensive assessment of ridley population dynamics, with emphasis on in-water life stages.
- Adopt an ecosystem-based approach to Kemp's ridley research and recovery efforts that includes all life history stages and integrates multiple systems of interest (i.e. nesting beach, pelagic, nearshore, abiotic and biotic).

- Increase collaboration among ridley researchers, as well as scientists from other disciplines, in order to gain a greater understanding of Kemp's ridley biology.
- Include consideration of Kemp's ridley habitat requirements in Integrated Coastal Zone Management along the US and Mexico coasts (Gibson *et al*., 1998; Douven et al., 2003). Kemp's ridleys may serve as a "focal" or "umbrella" species (Lambeck, 1997) through which other nearshore and estuarine-dependent species may be protected and managed.
- Continue long-term (10+ yrs), in-water surveys that are consistent and standardized for comparisons across years and regions. Furthermore,
    - Include more in-water study locations throughout Kemp's ridley's range.
    - Utilize more satellite tracking studies to ascertain Kemp's ridley long-term habitat use and associations.
- Examine Gulf of Mexico circulation patterns, mapped via models of sea surface height, to simulate possible hatchling dispersion paths and post-pelagic transport to benthic foraging grounds.
- Investigate the energetic requirements, feeding rates and prey size/type preferences of wild ridleys.
    - Continue to assess the impact of commercial fisheries (e.g. blue crab, shrimp, *etc*.) on ridley prey availability.

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APPENDIX A

Supplemental Figures

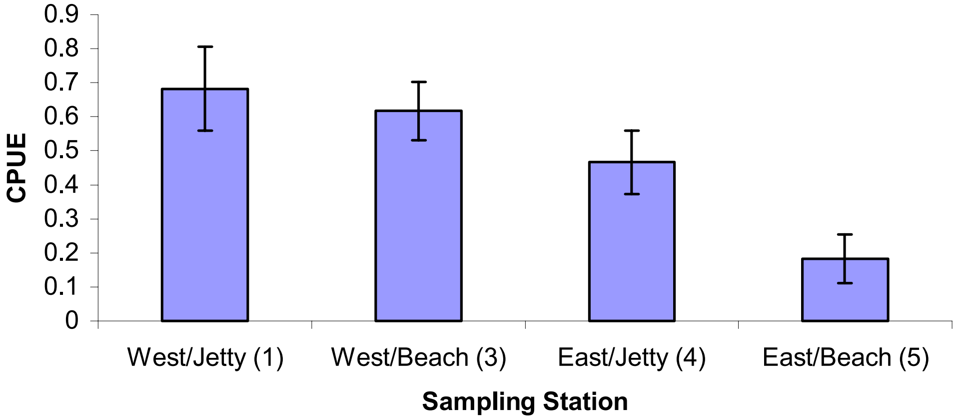


Figure A-1. Overall mean *L. kempii* CPUE (with standard error bars) for entanglement netting stations sampled at northwestern Gulf of Mexico sites during 1993-2002.

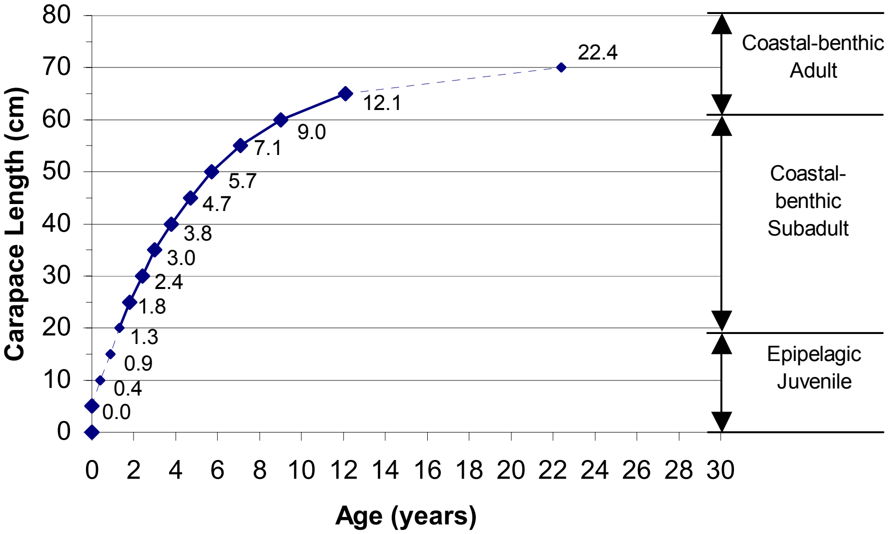
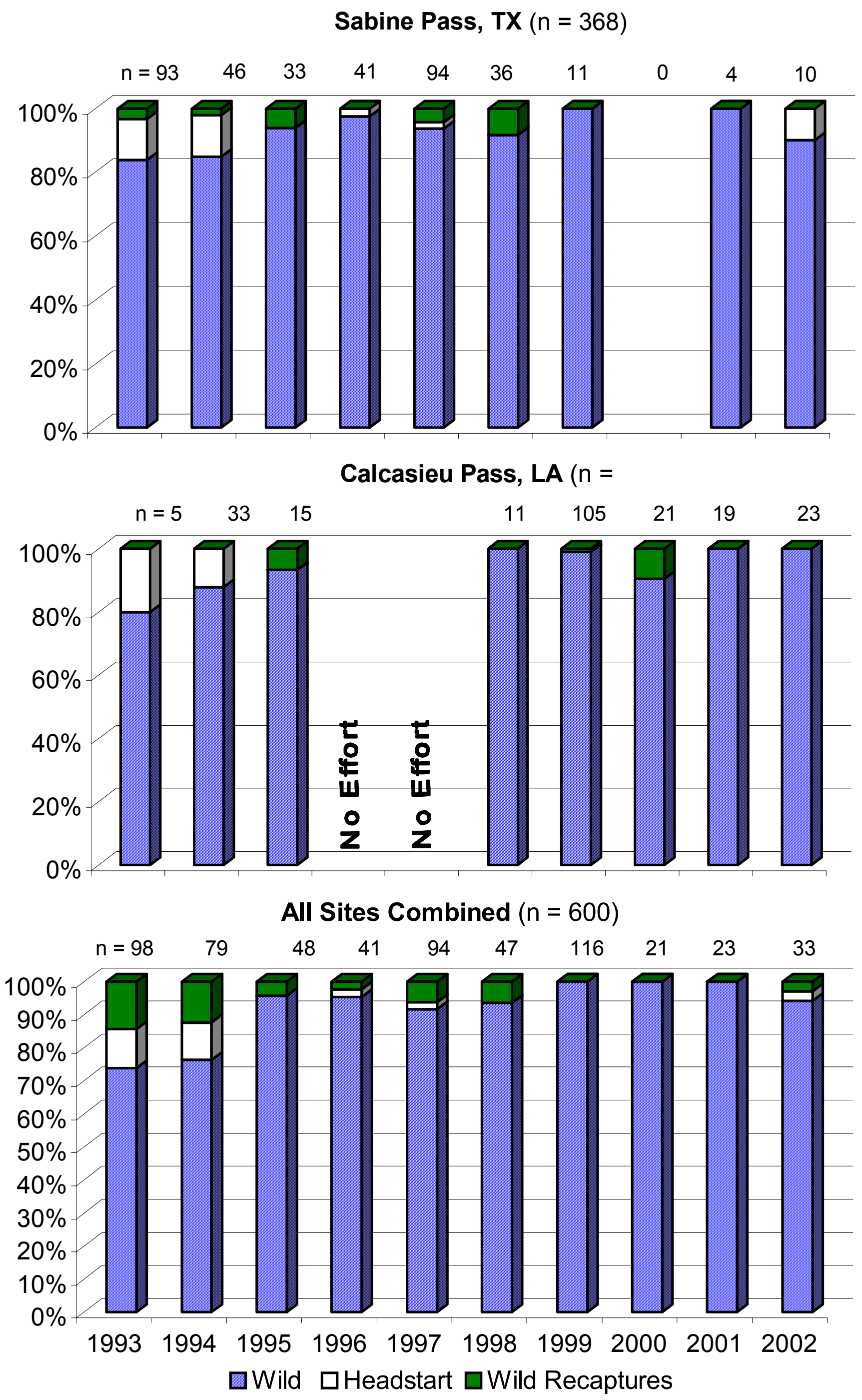


Figure A-2. Kemp's ridley growth curve associating size, age and life history stage. Dashed portion of curve represents extrapolation outside the size range of the database (adapted from TEWG, 2000).


Sabine Pass, TX (n = 368)

Calcasieu Pass, LA (n =

All Sites Combined (n = 600)

Figure A-3. Annual percent composition of wild, headstart and wild recaptured Kemp's ridleys at Sabine and Calcasieu Passes and all sites combined during 1993-2002 (n = total number of ridley captures) .

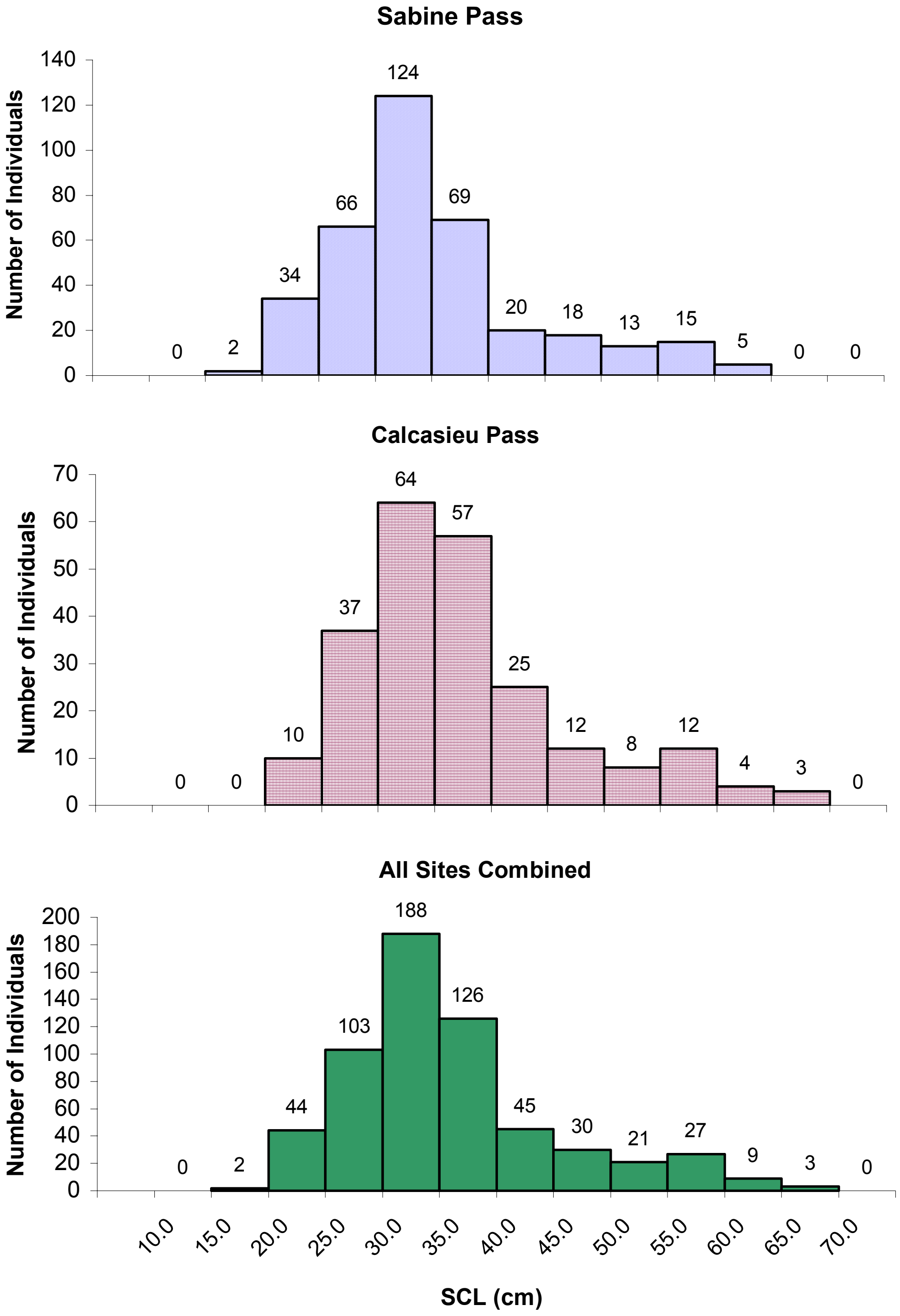


Figure A-4. Overall Carapace-length frequency for Kemp's ridleys captured at Sabine and Calcasieu Passes and all sites combined during 1993-2002.

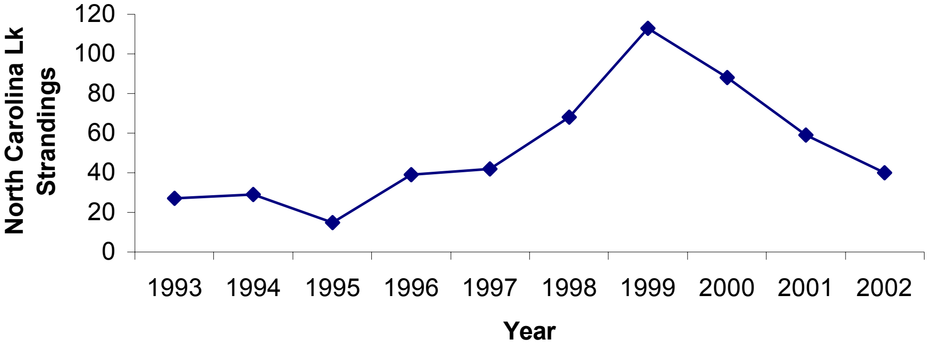


Figure A-5. Annual Kemp's ridley strandings in North Carolina during 1993-2002. (Data provided by M. Godfrey and W. Cluse of the NC Wildlife Resources Commission).

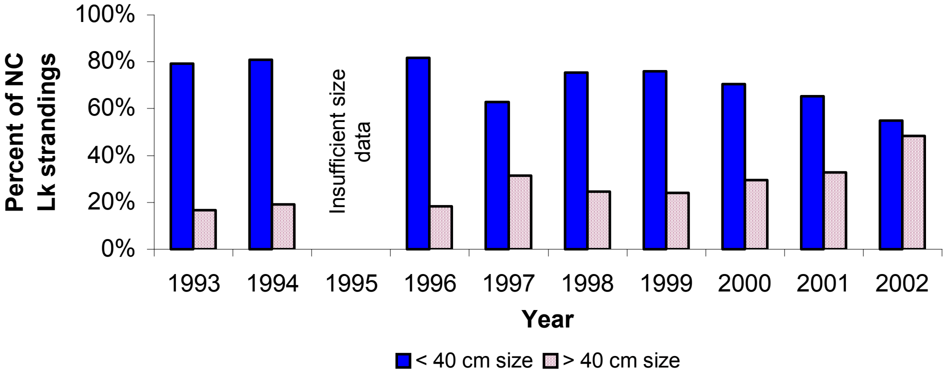


Figure A-6. Percentage of annual North Carolina ridley strandings with curved carapace length (CCL) or curved carapace width (CCW) less than and greater than 40 cm.

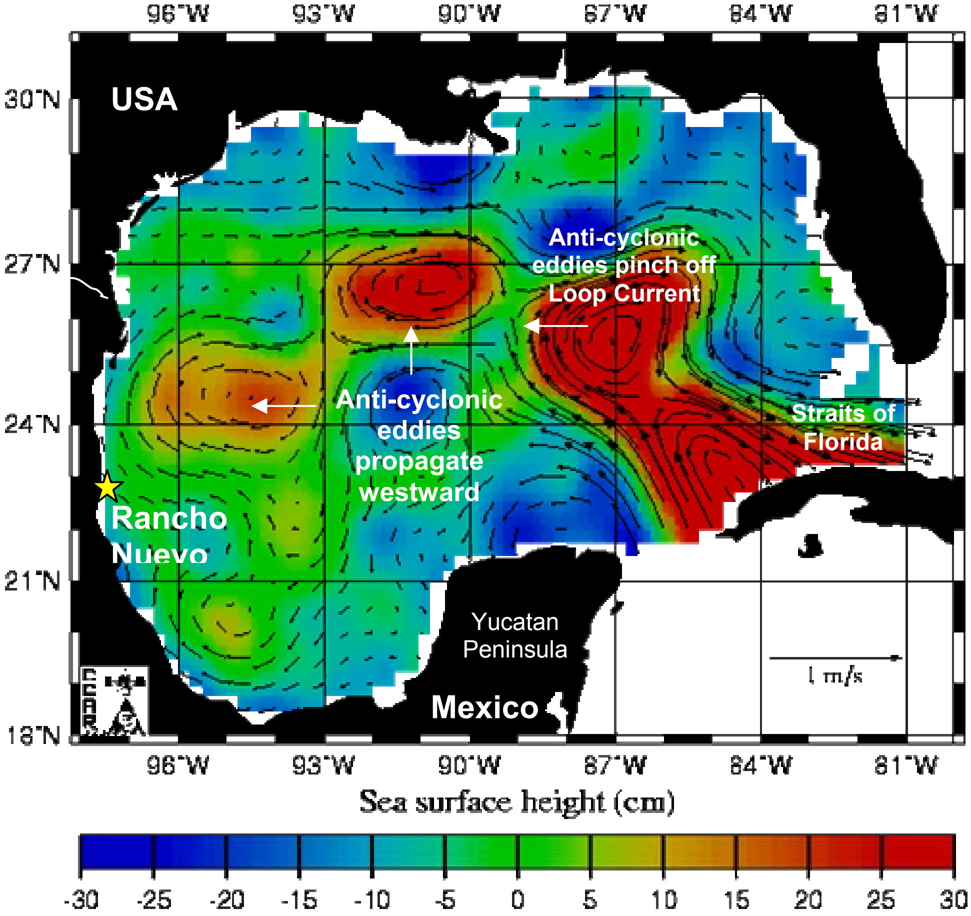


Figure A-7. Prominent Gulf of Mexico circulation features as mapped from satellite altimetry and sea surface height anomalies (TOPEX/ERS-1 Analysis, July 1995). Black arrows represent current velocity and direction of flow. Source: Colorado Center for Astrodynamics Research (CCAR).

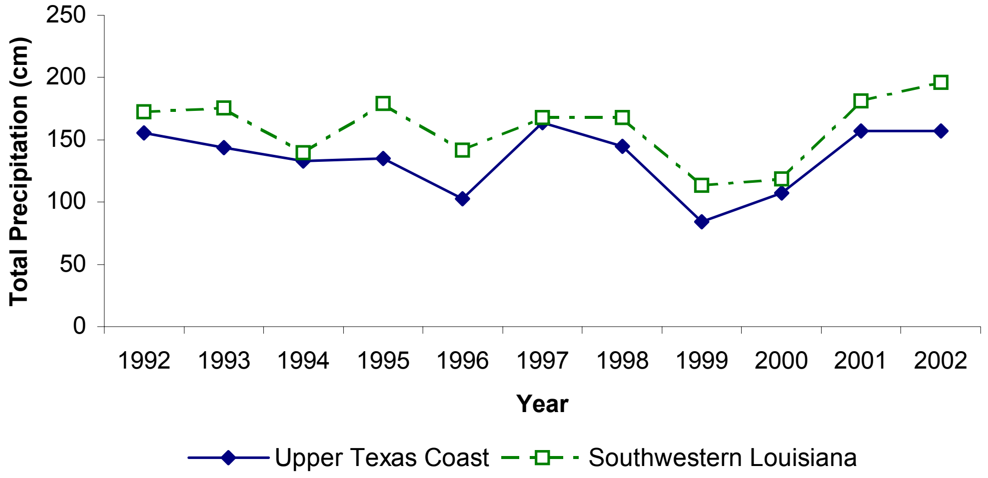


Figure A-8. Annual cummulative precipitation for the Upper Texas Coast and Southwestern Louisiana during 1992-2002. (Data Source: Western Regional Cimate Center website)

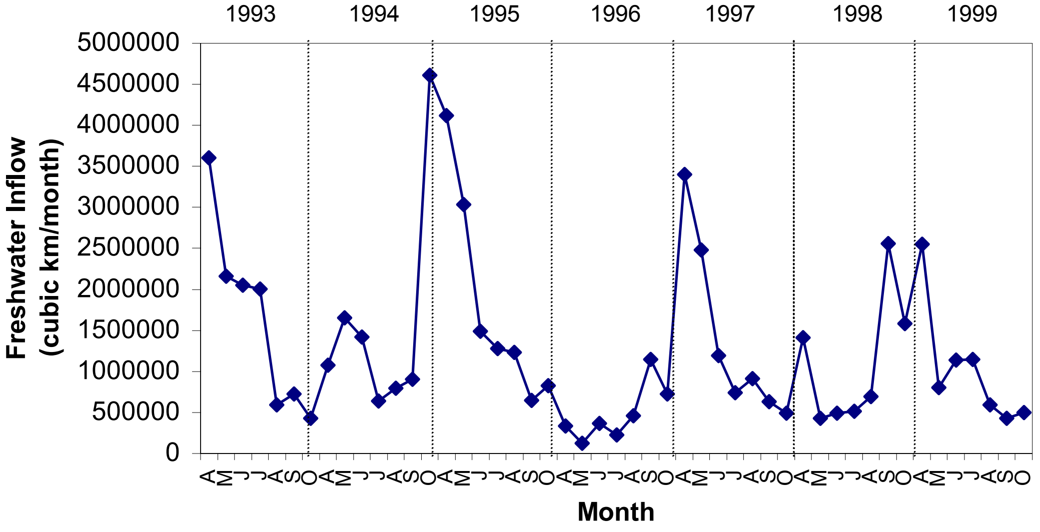


Figure A-9. Monthly freshwater inflow for Sabine Lake Estuary (April-October, 1993-1999). (Data Source: TWDB website)

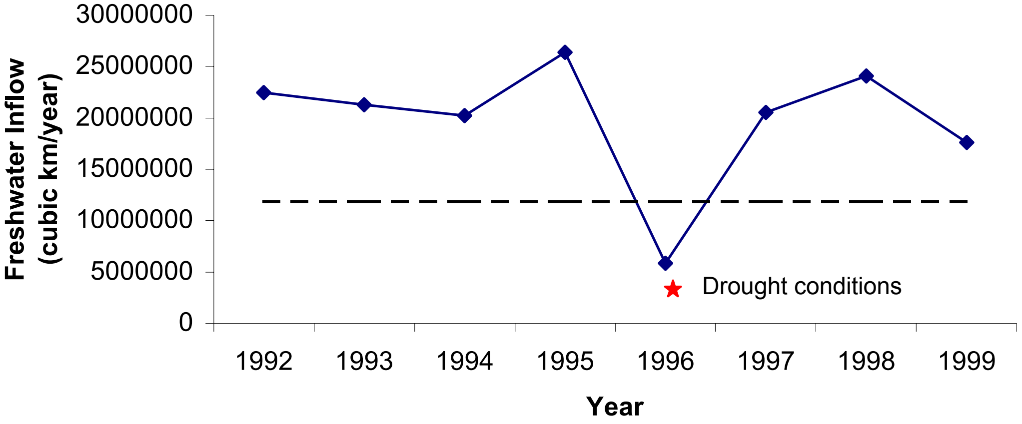


Figure A-10. Annnual freshwater inflow balance for the Sabine-Neches Estuary, 1992-1999. Optimal inflow needed (11.8 million cubic km/year) designated by dashed line. (Data source: TWDB website)

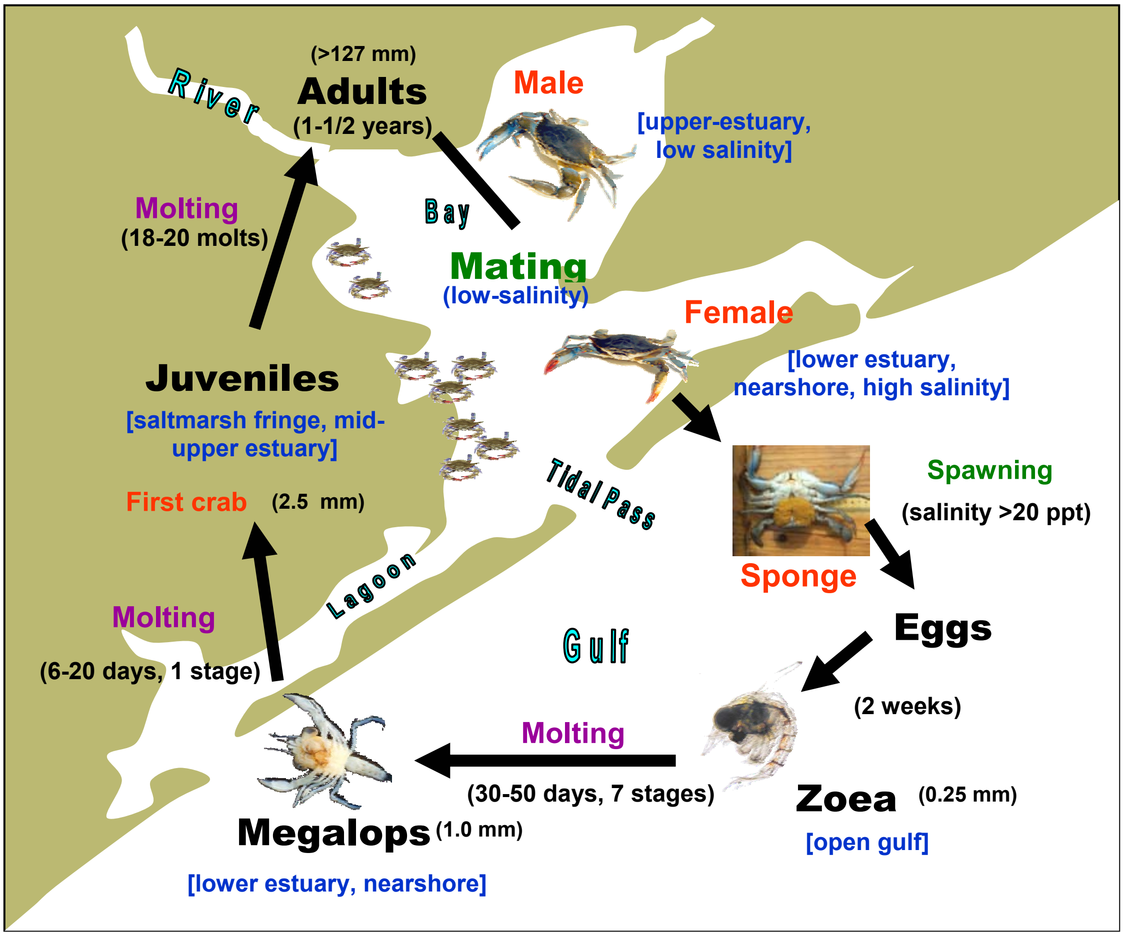


Figure A-11. Generalized blue crab life cycle showing representative life history stages and their typical location within an estuarine system and adjacent marine

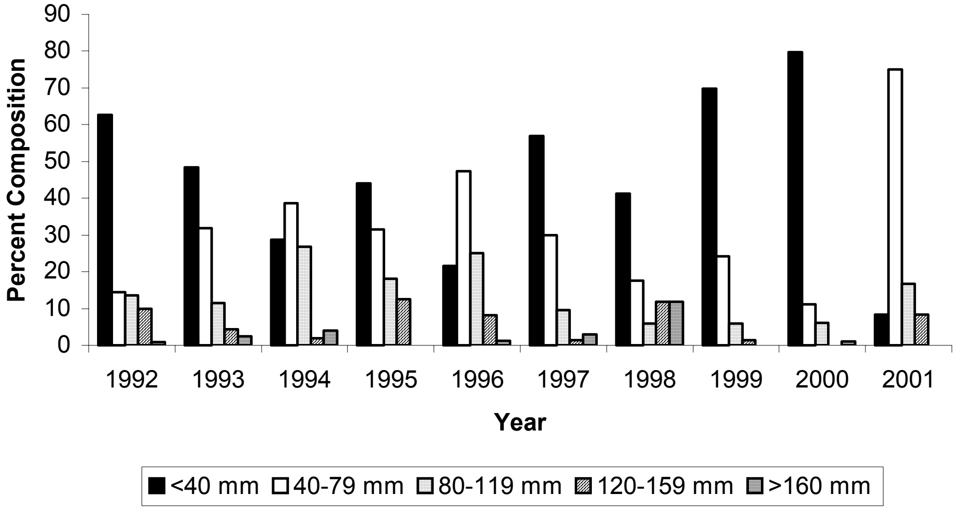


Figure A-12. Annual blue crab percent size composition from LDWF fishery-independent trawls collected in Gulf waters near Calcasieu Pass, LA (Data provided by V. Guillory) .

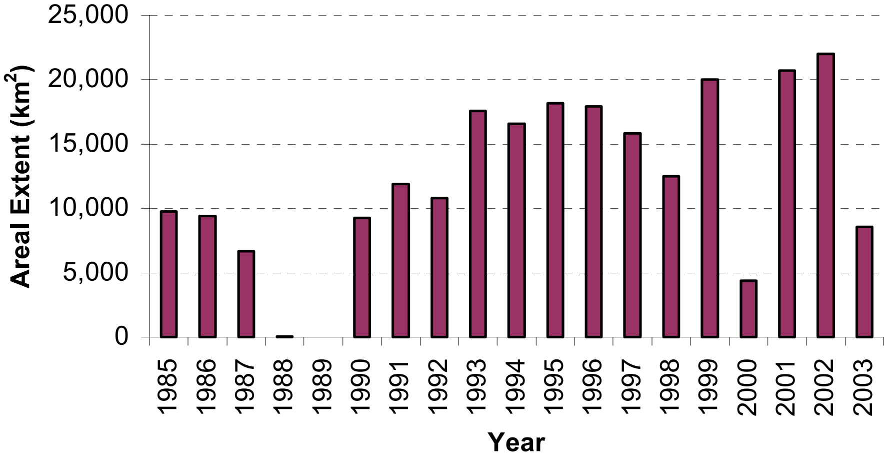


Figure A-13. Estimated areal extent of the hypoxic zone off the Louisiana coast. Data source: hypoxia studies of N.N. Rabalais, R.E. Turner and W.J. Wiseman, Jr.

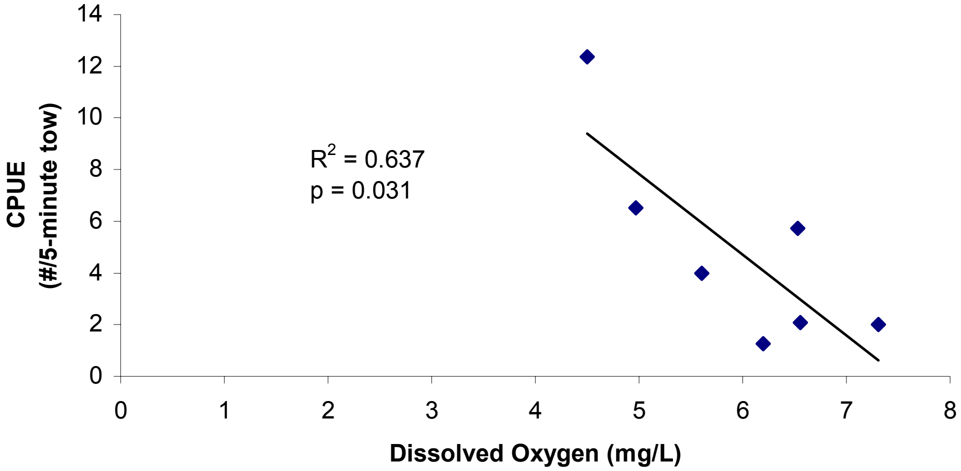


Figure A-14. Annual mean blue crab CPUE from trawl samples versus dissolved oxygen content at Sabine Pass during 1996-2002.

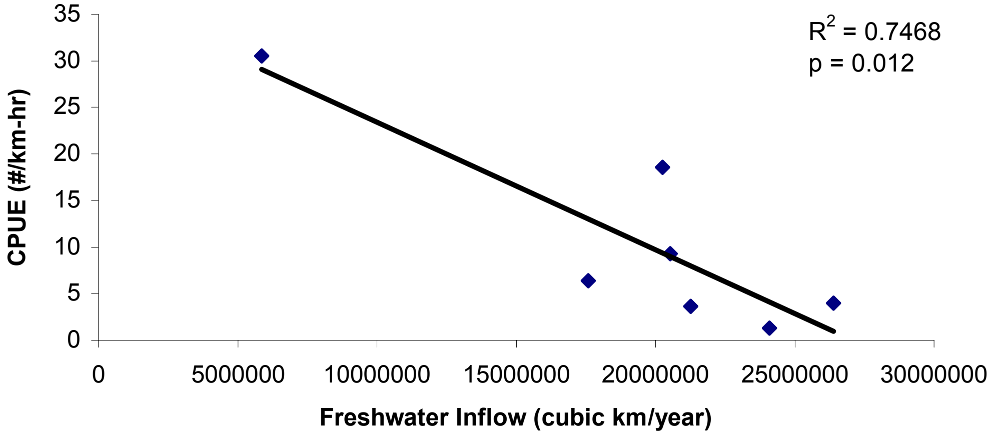


Figure A-15. Annual mean blue crab CPUE from entanglement nets at Sabine Pass versus freshwater inflow to Sabine Lake Estuary during 1993-1999 (Data source for freshwater inflow: TWDB website).

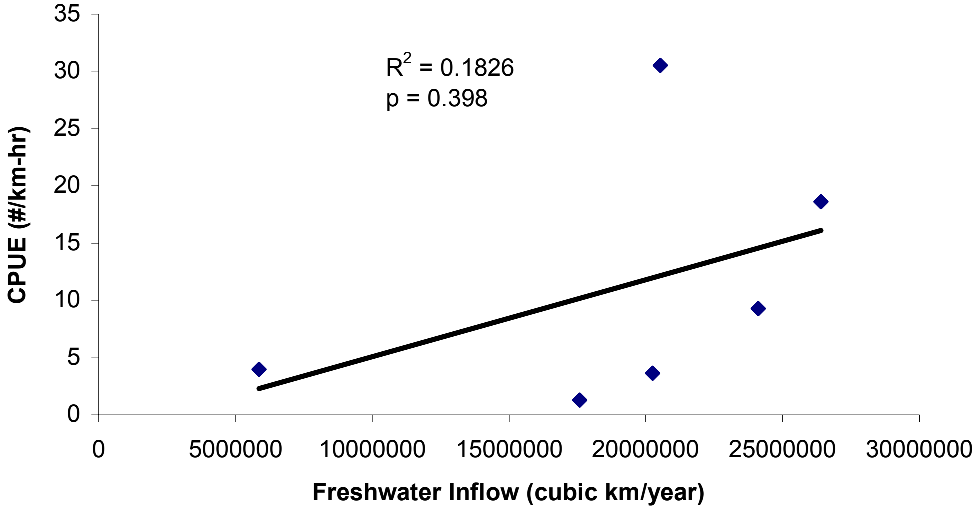


Figure A-16. Annual blue crab CPUE from entanglement nets at Sabine Pass versus freshwater inflow to Sabine Lake Estuary during 1993-1999, plotted with a one year lag (Data source for freshwater inflow: TWDB website).

APPENDIX B

Supplemental Tables

Table B-1. Sea turtle stranding totals along the Texas coast since 1994.

Year/Species	Cc	Lk	Cm	Dc	Ei	Un	Total
1994	194	255	48	3	14	14	528
1995	125	140	30	19	8	21	343
1996	202	123	119	11	41	14	510
1997	168	180	142	10	14	11	525
1998	169	132	59	6	16	14	396
1999	212	95	83	18	32	10	450
2000	163	100	90	14	28	17	412
2001	165	115	73	6	30	8	397
2002	112	97	45	17	45	8	324
Trendline Statistics							
p	0.381	0.041		0.528	0.064		0.123
y-intercept (b_o)	187.944	207.278		8.9722	10.1667		508.75
slope (b_1)	-4.0333	-13.967		0.5167	3.033		-15.417
r^2	0.111	0.552		0.059	0.409		0.305

Cc = *Caretta caretta*
Lk = *Lepidochelys kempii*
Cm = *Chelonia mydas*
Dc = *Dermochelys coriacea*
Ei = *Eretmochelys imbricata*
Un = Unknown

Table B-2. Monthly and annual entanglement netting effort (km-hours) at Sabine Pass, Calcasieu Pass, and all sites combined, during 1993-2002.

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	Total
Sabine Pass											
April	5.97	1.27	6.52	6.88	1.60						22.24
May	18.47	2.87	6.90	9.30	18.40	26.48	21.19	8.16	8.48		120.25
June	12.04	11.55	4.82	9.27	21.99	9.10	9.17	2.97	10.02	19.14	110.07
July	23.45	4.63	8.39	17.12	18.38	16.68	9.33	10.27	8.65	7.03	123.93
August	17.50	4.03	16.81	8.11	11.52	8.98	16.72	8.53	6.13		98.33
September	6.42	5.78	1.42	4.45	10.58						28.65
October	7.68	3.82	3.23	10.70							25.43
Total km-hours	**91.53**	**33.95**	**48.09**	**65.83**	**82.47**	**61.24**	**56.41**	**29.93**	**33.28**	**26.17**	**528.90**
Calcasieu Pass											
April											
May	1.26						9.37	3.55	2.50	4.87	21.55
June	4.24	2.63	1.50			8.40	11.26	21.28	23.62	21.74	94.67
July	6.24	11.92	10.06			8.83	11.63	16.00	12.15	20.62	97.45
August		20.78	8.03			6.98	17.22	12.48	23.48		88.97
September	3.74	2.88					37.60				44.22
October	1.23	5.07	4.88				21.02				32.20
Total km-hours	**15.48**	**17.43**	**11.56**			**17.23**	**79.61**	**37.21**	**38.27**	**47.23**	**264.02**
Over all NW Gulf											
April	5.97	1.27	6.52	6.88	1.60						22.24
May	19.73	2.87	6.90	9.30	18.40	26.48	30.56	11.71	10.98	4.87	141.80
June	16.28	14.18	6.32	9.27	21.99	17.50	20.43	24.25	33.64	40.88	204.74
July	29.69	16.55	18.45	17.12	18.38	25.51	20.96	26.27	20.80	27.65	221.38
August	17.50	24.81	24.84	8.11	11.52	15.96	33.94	21.01	29.61		187.30
September	10.16	8.66	1.42	4.45	10.58		37.60				72.87
October	8.91	8.89	8.11	10.70			21.02				57.63
Total km-hours	**107.01**	**51.38**	**59.65**	**65.83**	**82.47**	**78.47**	**136.02**	**67.14**	**71.55**	**73.40**	**792.92**

Blank = Not Sampled

km-hours = length of net x hours sampled / 1000 m

Effort shown for Calcasieu Pass in 1999 and 2000 also includes that from Mermentau Pass, LA.

VITA

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Tasha Lynn Metz was born on October 10, 1973 in Jefferson, Louisiana to Thomas B. and Janita D. Metz. Her family lived in Marrero, Louisiana and she attended high school at Louise S. McGehee School in New Orleans, where she graduated valedictorian of her class in June 1991. She entered Texas Christian University (Fort Worth, Texas) in August 1991 and graduated with a Bachelor of Science degree in biology in May 1995. She remained at Texas Christian University to receive her Master of Science degree in biology in May 1997. As a Master's student, she was a teaching assistant in the Biology Department and conducted research in Jamaica on littorine snails that inhabit the rocky shore. In October 1997, she was employed as an environmental scientist performing wetland delineations for Alan Plummer Associates, Inc. (Arlington, Texas). In August 1999, she entered graduate school at Texas A&M University to pursue a Doctor of Philosophy degree in wildlife and fisheries sciences. As a Ph.D. student, she was a teaching assistant in the Introductory Biology Program in College Station and in the Marine Biology Department in Galveston. In addition, she participated in sea turtle research projects, presenting results at scientific meetings, assisted in red snapper research, and was an instructor/field guide for the Sea Camp and Elderhostel Programs, as well as for the Galveston Graduate Student Association. Tasha received her Ph.D. in wildlife and fisheries sciences from Texas A&M University in August 2004.